CHAPTER 6

GENERALIZED TRAVEL COST MODEL: REVISED ESTIMATES

6.1 INTRODUCTION

This chapter offers a set of revised estimates for the site demand and generalized travel cost models developed in Desvousges, Smith, and McGivney [1983]. In particular, these revisions focus on three dimensions of the original analysis--the character of the data, the availability of the data, and the diversity of recreation activities undertaken at a site--and attempt to resolve the statistical problems imposed by them. In addition to introducing the role of activities in the models, the following sections extend significantly the treatment of several statistical issues raised in Chapter 7 of Desvousges, Smith, and McGivney [1983].

The generalized travel cost models are estimated in two stages. First, individual travel cost demand equations are estimated for each of the 43 Corps of Engineers sites. The demand equations estimated using ordinary least squares (OLS) are compared with those using the maximum likelihood (ML) estimator for the censored and truncated dependent variable. Second, the parameters and standard errors from the first stage are regressed, using a generalized least-squares approach (GLS) on the site characteristics and activity variables. The results of these procedures are evaluated for their sensitivity to both the sample of sites used and the model specifications. This sensitivity is crucial because neither set of estimates can be viewed as the "truth." The ML approach also involves judgments, albeit different ones than OLS.

Specifically, Section 6.2 outlines the statistical problems in using OLS with truncated and censored data, defines the ML estimator used to deal with these problems, and discusses its relationship to OLS, as well as to other more rigorous estimators. Section 6.3 reports the ML estimates and compares them with the OLS estimates for each site. Section 6.4 describes the results of the various extensions of the analysis, including the addition of new data for previously unconsidered variables and the introduction of various mixes of specific recreation activities. Finally, the main points are summarized in Section 6.5.

6.2 THE GENERALIZED TRAVEL COST MODEL WITH TRUNCATED AND CENSORED DATA

This section examines the estimation of the travel cost model using OLS when the dependent variable is truncated and censored, explores the estimation implications of these problems, and proposes an ML estimator to deal with them. It provides an historical perspective while it also considers a few methodological issues that arise in choosing the appropriate estimator.

6.2.1 OLS Estimation

The first dimension of the original analysis that is problematic for the generalized travel cost model is the character of the data available for its application. In particular, the available data on the dependent variable, visits, are truncated at one because the survey was conducted onsite. The omission of individuals who chose not to visit the site implies that the model is unable to reflect the reservation or choke price--i.e., the price at which visits to the site drop off to zero. It also means that the model will be unable to describe the behavior of those individuals if they differ significantly from the site visitors. The importance of these problems arises when the model is used to estimate benefits of water quality improvements. For example, the lack of data on the reservation price requires a judgment on a suitable proxy for the benefits calculations. In addition, truncation creates statistical problems for the use of

OLS. In particular, it biases the estimates of the travel cost demand functions for the 43 sites. Additional estimation difficulties arise because the coding procedure for visits censored the number of visits at the upper end. Both truncation and censoring create problems similar to specification errors in econometric analysis. In effect, truncation and censoring of the dependent variable affects the properties of the residuals from the estimated equation in the ways similar to misspecifying (e.g., omitting a key variable) the equation itself. The residuals diverge from the assumptions necessary for OLS to be judged the best linear unbiased estimator.

The importance of the two problems for the generalized travel cost model parallels the increasing recognition in econometrics of the effects of the features of the sampling procedures underlying the survey data, and the transformations applied to key variables for econometric estimators. This recognition can be traced to early work by Tobin [1958] but did not have an appreciable impact on econometric analysis until the mid-seventies. One reason for the delay can be found in the absence of detailed micro datasets. Moreover, when they were available, they often were collected in response to noneconomic objectives and, thus, omitted important economic variables. Consequently, empirical models based on these data were regarded as crude proxy relationships with acknowledged specification errors. In such a context, there is little incentive for refining estimation procedures to remove any one of the many sources of bias in the estimated parameters.

Examples of this reasoning are readily found in the literature on outdoor recreation. Although early household surveys associated with national recreation plans provided micro data sets, they rarely included all the variables important in economic models of behavioral decisions. Cicchetti, Seneca, and

Davidson [1969], for example, considered probit and tobit estimators, but did not report the results because they felt the OLS results agreed in sign and significance with the more appropriate estimators. They felt that any improvement with other methods did not outweigh the increased estimation costs. For example, in discussing their selection of OLS (referred to as "classical least squares" in their discussion), Cicchetti, Seneca, and Davidson [1969, p. 86] observed that:

> Costs are a prime consideration with probit, since errors in specification may never be anticipated as they can be in classical least squares; thus each alternative specification has an initial new set-up cost as well as an estimation cost. These cost considerations normally favor classical least squares over both alternatives [i.e., generalized least squares and probit].

However, improvements in data sets and computational capabilities have shifted the balance toward more rigorous investigations. Yet, even today, maximum likelihood estimators that account for the effects of sampling procedures or variable categorizations can be expensive. Consequently, a role still exists for OLS in general estimation strategies applied to these cases. Thus, the OLS results from our initial investigations are reported along with their more rigorous ML counterparts.

6.2.2 Truncated Data

Truncation is the first aspect of limited, dependent variable models relevant to the measures of site use (visits) available in the Federal Estate Survey. It arises because the survey was an onsite survey. Truncation can bias the OLS estimates of the site demand models. An example adapted from Maddala [1983] illustrates this point. Suppose a simple travel cost demand model with the log of visits (our measure of site usage) is specified to be a function of travel costs (including vehicle and time costs) as in Equation (6.1),

$$\ln V_i = \alpha_0 + \alpha_1 TC_i + \varepsilon_i \tag{6.1}$$

where

V_i = visits to the site during the season by i^{th} individual

TC_i = travel costs per trip for the i^{th} individual

ε_i = stochastic error for the i^{th} individual.

The assumption that ε_i follows a standard normal distribution, with an expected value of zero and a variance of one, simplifies the description of the properties of the truncated error (see Johnson and Kotz [1970] for details). Normality is implicitly used in the statistical inference of models based on OLS. Truncation affects the sampling process and, in turn, affects the estimated models that can be used with data elicited from visitors to each site. Only V_i and TC_i for the sample individuals are observed, implying that V_i must be at least one in order for the survey to have information on the demand and travel costs. For the model, this implies that ε_i is constrained by the sampling process. Equation (6.2) suggests that the condition in $V_i > 0$ implies a constraint on ε_i.

$\ln V_i \geq 0$?

Can't it equal zero? $\ln(1) = 0$.

$$\varepsilon_i > - \alpha_0 - \alpha_1 TC_i . \tag{6.2}$$

$\geq$ hence

Thus, while our error is assumed to follow a standard normal distribution, what is observed is truncated and cannot have a zero expectation.* The expected value will be a function of the independent variables in the model, as given in Equation (6.3).

$$E(\tilde{\varepsilon}_i) = \frac{\phi(-\alpha_0 - \alpha_1 TC_i)}{1 - \Phi(-\alpha_0 - \alpha_1 TC_i)} \tag{6.3}$$

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*See Maddala [1983] for further discussion.

where

$\phi(\cdot)$ = density function for the standard normal

$\Phi(\cdot)$ = distribution function for the standard normal*

$\tilde{\varepsilon}_i$ = the truncated version of ε_i with truncation from below at $-\alpha_0 - \alpha_1 \, TC_i$.

As a result, it cannot be assumed that the error, $\tilde{\varepsilon}_i$, will be independent of the specified determinants of demand. Heckman [1976, 1979] has paralleled this problem with those from specification errors. Both lead to non-null expectations for the errors and can lead to biased estimates of the parameters for the included independent variables. Only in the simplest cases can one describe the nature of the bias a priori. Heckman [1976, pp. 476-478] observed that:

> All of the models in the literature developed for limited dependent variables and sample selection bias may be interpreted within a missing data framework . . . . The bias that arises from using least squares to fit models for limited dependent variables or models with censoring or truncation arises solely because the condition mean of V_{1i} [the error] is not included as a regressor. The bias that arises from truncation or selection may be interpreted as arising from an ordinary specification error with the conditional mean deleted as an explanatory variable. In general, one cannot sign the direction of bias that arises from omitting this conditional mean.

Thus, empirical judgments cannot be avoided in any attempt to understand the effects of how these problems should be treated in estimating a model. The exact nature of these judgments, and their importance in benefits estimation, is explored throughout this chapter and Chapter 7.

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*The specific forms for the standard normal density and distribution functions are given as:

$$\phi(Z) = \frac{1}{\sqrt{2\pi}} \exp\left(-\frac{1}{2} Z^2\right)$$

$$\Phi(Z) = \frac{1}{\sqrt{2\pi}} \int_{-\infty}^{Z} \exp\left(-\frac{1}{2} x^2\right) dx$$

$$\phi(Z) = \frac{d\Phi(Z)}{dZ} .$$

6.2.3 Censored Data

Censoring problems in survey data exert influences similar to truncation of the dependent variable. The problem arises because the visit information was reported with the last interval open-ended. This limits the amount of behavioral information available on some observations of recreationists. Censoring implies that the values for those individuals who visited the site six times cannot be distinguished from those who visited more frequently. Thus, if $\ln \tilde{V}$ is the censoring point, and ε_i is assumed to follow a normal (ignoring the truncation problems), the density function, g(•), for the censored variable is given as follows:

$$g(\cdot) = \begin{cases} \frac{1}{\sigma}\phi\left(\frac{\ln V_i - \alpha_0 - \alpha_1 TC_i}{\sigma}\right) & \text{for } \ln V_i < \ln \tilde{V} \\ 1 - \Phi\left(\frac{\ln \tilde{V} - \alpha_0 - \alpha_1 TC_i}{\sigma}\right) & \text{for } \ln V_i \geq \ln \tilde{V} . \end{cases} \tag{6.4}$$

Ignoring this problem would also lead to a nonnull expectation for the error and would, to some extent, be reflected in the Olsen [1980] index. This index served as a guide for identifying sites with potential truncation problems in the earlier work in Desvousges, Smith, and McGivney [1983]. However, it is unknown how powerful this index is because this application exceeds the theoretical basis of the Olsen approximation.

The importance of both censoring and truncation problems to the OLS estimates will depend on the dispersion of the measure of site usage. If the number of visits tends to cluster within the bounds imposed by the truncation and the censoring points, one would expect little effect on the OLS parameter estimates. By contrast, if the sample includes a larger portion of the observations at these points, then the effects should be pronounced. Since this logic lies at the heart of Olsen's approximation and the application of it as a gauge of the severity of these effects, our use of the approximation to screen sites for truncation/censoring effects may have been sufficient to mitigate the effects of the error structure on the final estimates of the generalized travel cost model. Of course, until the actual estimation was completed using the ML estimator, our decision was simply a judgment based on diagnostic indexes. The results reported in Section 6.3 indicate that the judgment was not uniformly appropriate for all sites.

However, before turning to a description of the ML estimator used to derive these results and to the estimates themselves, it is important to provide some perspective on them. Estimators that take into account truncation and/or censoring effects assume a specific error distribution. For models involving selectivity effects, Goldberger [1980] has shown that their results can be sensitive to the specified errors. The common structure of selectivity, truncation, censoring, and misspecification problems make his conclusion relevant to all of these issues.

This sensitivity has, in turn, fostered a substantial amount of research [see, for example, Arabmayer and Schmidt, 1982; Olsen, 1982; and Lee, 1984]. Indeed, in concluding his analysis of a test for selectivity effects based on testing for shifts in the least-squares residuals distribution, Olsen [1982, p. 236] observed that:

> In assessing this and other work on selectivity bias we repeat that in the absence of some structure the problem of correcting for selectivity bias cannot be solved. The identification problem is omnipresent . . . . We have shown here that maximum likelihood methods have the little appreciated attribute that they are extremely sensitive to the assumption made about the population distribution of the regression residuals.

What all of this implies is that the new estimates are not to be interpreted as the "truth" and the earlier OLS estimates as invalid. Both sets of estimates result from organizing sample information based on maintained assumptions. In real-world applications, true parameter values are unknown, and no set of maintained assumptions underlying one estimator is uniformly superior to another set. In effect, increased complexity in estimation does not relieve the need for judgment. Rather, the maximum likelihood estimator provides another basis for judging the sensitivity of the demand models' estimates to the treatment of censoring and truncation problems. This sensitivity analysis will add another ingredient to the recipe for examining how sensitive the estimated benefits of water quality improvements are to the models used to derive them.

To turn to the definition, the ML estimator is assumed to have a normal error with zero expectation and variance σ^2. For a model truncated at zero (because of the semi-log transformation) and censored at the maximum number of recorded trips (k), the density function for the error of the ML estimator can be defined as in Equation (6.5).

$$f(\varepsilon_i) = \begin{cases} \dfrac{\frac{1}{\sigma}\phi\,[(\ln V_i - \alpha_0 - \alpha_1 TC_i)/\sigma]}{1 - \Phi[(-\alpha_0 - \alpha_1 TC_i)/\sigma]} \\[2ex] \text{for } k > \ln V_i \geq 0 \\[2ex] \dfrac{1 - \Phi[(k - \alpha_0 - \alpha_1 TC_i)/\sigma]}{1 - \Phi[(-\alpha_0 - \alpha_1 TC_i)/\sigma]} \\[2ex] \text{for } \ln V_i \geq k\ . \end{cases} \qquad (6.5)$$

The likelihood function for this model is then

$$L(\alpha_0, \alpha_1, \sigma^2 \ln V_1, \ldots, \ln V_n) =$$

$$\prod_{0 \leq \ln V_i < k} \left[\frac{\frac{1}{\sigma} \phi \, [(\ln V_i - \alpha_0 - \alpha_1 TC_i)/\sigma]}{(1 - \Phi[(-\alpha_0 - \alpha_1 TC_i)/\sigma])} \right] \times \tag{6.6}$$

$$\prod_{\ln V_i \geq k} \left[\frac{1 - \Phi[(k-\alpha_0 - \alpha_1 TC_i)/\sigma]}{1 - \Phi[(-\alpha_0 - \alpha_1 TC_i)/\sigma]} \right].$$

Estimates of α_0, α_1, and σ are derived by numerically maximizing the log-likelihood function using a variation on the Davidon, Fletcher, Powell [1963] algorithm available in GQOPT (the set of numerical optimization procedures and software developed by Stephen Goldfeld and Richard Quandt and provided by the Econometric Research Program at Princeton University).

6.3 REVISED ESTIMATES: A COMPARISON BETWEEN THE OLS AND THE ML ESTIMATES OF THE SITE DEMAND MODELS

After the generalized travel cost model was revised to accommodate the truncated and censored data, the first stage of the two-stage model was estimated to obtain new demand functions for all 43 of the recreation sites encompassed by the survey. This section reports these new functions and compares them with the values that were presented in Desvousges, Smith, and McGivney [1983] using the OLS estimator.

Table 6-1 reports the new ML estimates of demand functions for all 43 sites using the simple general model format with the log of visits specified to be a linear function of roundtrip travel costs (including vehicle-related transportation costs and the time costs of travel, valued as before using the projected wage rate). On Table 6-1, the ML estimates and the ratios of these parameter estimates to their asymptotic standard errors are listed above the original OLS estimates (labeled OLS-I) taken from Table 7-4 in Desvousges, Smith,

Table 6-1. Maximum Likelihood and OLS Estimates of General Model by Site
LN Visits = $\alpha_0 + \alpha_1$ (T+M) Costs + α_2 Income

Site name	Site No.	Estimator	Intercept	T+M cost	Income	Function value	R^2	df
Allegheny River System, PA	300	ML	-35.95 (-80.93)	0.0311 (40.16)	4.4×10^{-4} (214.59)	13.92	-	-
		OLS-I	0.53 (2.04)	-0.0005 (-0.13)	8.2×10^{-6} (0.74)	-	0.01	66
Arkabutla, Lake, MS	301	ML	2.33 (8.21)	-0.0473 (-6.20)	1.9×10^{-6} (0.11)	-24.00	-	-
		OLS-I	1.58 (9.99)	-0.0093 (-3.09)	6.2×10^{-6} (0.67)	-	0.15	58
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	ML	2.31 (2.31)	-0.0125 (-0.28)	1.6×10^{-5} (64.95)	-17.67		
		OLS-I	2.31 (9.76)	-0.0125 (-2.30)	-1.8×10^{-5} (-1.08)	-	0.14	38
Beaver Lake, AR	303	ML	2.23 (10.46)	-0.0216 (-12.50)	-5.5×10^{-6} (-0.74)	57.54	-	-
		OLS-I	1.61 (16.07)	-0.0066 (-12.77)	-3.5×10^{-6} (-0.78)	-	0.43	224
Belton Lake, TX	304	ML	2.94 (4.62)	-0.0727 (-2.70)	1.2×10^{-5} (0.42)	-23.61	-	-
		OLS-I	1.69 (9.38)	-0.0052 (-2.47)	2.6×10^{-6} (0.29)	-	0.12	50
Benbrook Lake, TX	305	ML	2.45 (1.54)	-0.0472 (-1.09)	8.3×10^{-5} (0.60)	-16.01	-	-
		OLS-I	1.83 (10.70)	-0.0054 (-4.11)	6.0×10^{-6} (0.80)	-	0.30	43
Berlin Reservoir, OH	306	ML	1.40 (1.40)	0.0014 (0.03)	2.3×10^{-6} (18.04)	-60.77	-	-
		OLS-I	1.40 (8.47)	0.0014 (0.43)	-4.1×10^{-7} (-0.05)	-	0.01	93
Blakely Mt. Dam, Lake Ouachita, AR	307	ML	2.44 (24.03)	-0.0374 (-13.63)	-9.6×10^{-6} (-0.88)	-18.17	-	-
		OLS-I	1.70 (10.08)	-0.0079 (-5.14)	-7.6×10^{-6} (-0.98)	-	0.24	88
Canton Lake, OK	308	ML	3.96 (8.94)	-0.2788 (-12.50)	1.4×10^{-4} (11.23)	-12.51	-	-
		OLS-I	1.77 (8.61)	-0.0206 (-5.28)	7.1×10^{-6} (0.86)	-	0.28	71

(continued)

Table 6-1 (continued)

Site name	Site No.	Estimator	Intercept	T+M cost	income	Function value	R^2	df
Clearwater Lake, MO	309	ML	0.10 (0.32)	-0.0620 (-13.82)	-1.3×10^{-4} (-31.89)	-20.15	-	-
		OLS-I	1.51 (5.97)	-0.0032 (-1.42)	-1.0×10^{-5} (-1.21)	-	0.04	71
Cordell Hull Dam and Reservoir, TN	310	ML	2.91 (87.61)	-0.0657 (-22.02)	3.8×10^{-6} (0.90)	-29.26	-	-
		OLS-I	1.86 (14.13)	-0.0139 (-6.00)	-1.2×10^{-8} (-0.01)	-	0.34	101
DeGray Lake, AR	311	ML	2.36 (3.55)	-0.0267 (-1.57)	-1.5×10^{-5} (-0.56)	-17.81	-	-
		OLS-I	1.79 (7.71)	-0.0070 (-3.00)	-6.9×10^{-5} (-0.73)	-	0.17	46
Dewey Lake, AR	312	ML	-0.48 (-27.37)	-0.0127 (-32.21)	5.7×10^{-5} (8.68)	27.44	-	-
		OLS-I	0.42 (2.27)	-0.0024 (-2.95)	2.0×10^{-5} (2.02)	-	0.18	43
Ft. Randall, Lake Francis Case, SD	313	ML	1.32 (1.32)	-0.0328 (-0.03)	6.2×10^{-5} (0.02)	21.14	-	-
		OLS-I	1.32 (6.00)	-0.0066 (-5.93)	7.5×10^{-6} (0.91)	-	0.43	47
Grapevine Lake, TX	314	ML	2.71 (6.41)	-0.0311 (-3.43)	1.8×10^{-5} (1.42)	-26.92	-	-
		OLS-I	1.80 (16.12)	-0.0073 (-8.80)	8.5×10^{-6} (1.70)	-	0.47	39
Greers Ferry Lake, AR	315	ML	2.10 (15.91)	-0.0287 (-9.84)	2.8×10^{-5} (3.20)	-51.84	-	-
		OLS-I	1.48 (14.08)	-0.0065 (-9.02)	8.4×10^{-6} (1.42)	-	0.28	[illegible]
Grenada Lake, MS	316	ML	4.92 (8.97)	-0.0924 (-4.58)	-3.5×10^{-5} (-0.58)	-29.47	-	-
		OLS-I	2.04 (12.61)	-0.0095 (-4.36)	-1.0×10^{-5} (-0.68)	-	0.22	[illegible]
Hords Creek Lake, TX	317	ML	2.77 (5.07)	-0.0502 (-2.38)	-6.5×10^{-5} (-2.22)	-13.49	-	-
		OLS-I	1.73 (8.22)	-0.0050 (-2.11)	-2.1×10^{-5} (-1.76)	-	0.19	[illegible]

(continued)

Table 6-1 (continued)

Site name	Site No.	Estimator	Intercept	T+M cost	Income	Function value	R^2	df
Isabella Lake, CA	318	ML	1.36 (3.25)	-0.0648 (-9.26)	3.6×10^{-5} (1.93)	-0 21	-	-
		OLS-I	1.26 (5.55)	-0.0073 (-3.15)	7.9×10^{-6} (0.81)	-	0.20	45
Lake Okeechobee and Waterway, FL	319	ML	5.63 (6.64)	-0.3489 (-6.88)	5.3×10^{-5} (1.77)	-6.54	-	-
		OLS-I	1.68 (3.68)	-0.0268 (-1.72)	1.9×10^{-7} (0.01)	-	0 10	27
Lake Washington Ship Canal, WA	320	ML	-0.04 (-0.04)	-0.0226 (-10.40)	7.7×10^{-5} (1.63)	3 76	-	-
		OLS-I	0.96 (2.69)	-0.0037 (-3.79)	1.7×10^{-5} (0.84)	-	0.26	41
Leech Lake, MN	321	ML	-2.57 (-5.45)	-0.0292 (-5.55)	6.1×10^{-5} (4.54)	13.67	-	-
		OLS-I	0.87 (3.88)	-0.0022 (-1.83)	3.5×10^{-6} (0.37)	-	0.07	45
Melvern Lake, KS	322	ML	-2.42 (-2.19)	-0.1797 (-20.00)	7.4×10^{-5} (2.56)	-14.17	-	-
		OLS-I	1.30 (4.47)	-0.0079 (-1.66)	4.1×10^{-6} (0.32)	-	0.06	42
Millwood Lake, AR	323	ML	1.43 (2.97)	-0.0331 (-6.15)	7.4×10^{-5} (2.97)	-20.14	-	-
		OLS-I	1.43 (7.94)	-0.0081 (-3.99)	1.8×10^{-5} (2.14)	-	0.25	50
Mississippi River Pool No. 3, MN	324	ML	1.28 (458.98)	-0.0319 (-96.36)	2.4×10^{-5} (6.19)	9.90	-	-
		OLS-I	1.33 (4.20)	-0.0057 (-4.62)	4.7×10^{-6} (0.54)	-	0.34	46
Mississippi River Pool No. 6, MN	325	ML	1.49 (2.67)	-0.0565 (-1.75)	5.8×10^{-5} (1.41)	-22.21	-	-
		OLS-I	1.41 (7.45)	-0.0074 (-4.39)	1.3×10^{-5} (1 53)	-	0.22	68
Navarro Mills Lake, TX	327	ML	2.09 (12.85)	-0.0909 (-5.91)	-1.4×10^{-4} (-9.29)	-16.34	-	-
		OLS-I	1.66 (6.40)	-0.0057 (-1.39)	-1.4×10^{-5} (-1.14)	-	0.06	39

(continued)

Table 6-1 (continued)

Site name	Site No.	Estimator	Intercept	T+M cost	Income	Function value	R^2	df
New Hogan Lake, CA	328	ML	-59.98 (-13.60)	-0.1342 (-3.26)	8.5×10^{-4} (13.71)	-11.60	-	-
		OLS-I	1.04 (2.58)	-0.0040 (-0.41)	7.1×10^{-6} (0.60)	-	0.01	38
New Savannah Bluff Lock & Dam, GA	329	ML	3.28 (2.24)	-0.0538 (-0.68)	-5.6×10^{-5} (-0.59)	-19.51	-	-
		OLS-I	1.88 (8.39)	-0.0067 (-1.44)	-9.8×10^{-6} (-0.70)	-	0.06	36
Norfork Lake, AR	330	ML	0.11 (30.69)	-0.0440 (-2003.36)	7.8×10^{-5} (616.05)	3.77	-	-
		OLS-I	1.13 (4.27)	-0.0047 (-2.55)	9.3×10^{-5} (0.79)	-	0.14	39
Ozark Lake, AR	331	ML	1.98 (3.70)	-0.0230 (-14.25)	1.2×10^{-5} (0.36)	-8.27	-	-
		OLS-I	1.66 (8.52)	-0.0046 (-4.44)	-8.8×10^{-6} (0.66)	-	0.31	[illegible]
Perry Lake, KS	332	ML	1.61 (2.52)	-0.0094 (-0.79)	-2.0×10^{-5} (-0.69)	-7.18	-	
		OLS-I	1.50 (4.17)	-0.0042 (-0.74)	-1.0×10^{-5} (-0.68)	-	0.03	[illegible]
Philpott Lake, VA	333	ML	2.21 (4.77)	-0.0335 (-22.71)	2.2×10^{-5} (0.80)	-8.80	-	-
		OLS-I	1.90 (9.28)	-0.0087 (-4.40)	-1.7×10^{-6} (-0.13)	-	0.36	[illegible]
Pine River, MN	334	ML	-1.74 (-795.49)	-0.0308 (-530.60)	-8.7×10^{-5} (-1075.39)	38.71	-	-
		OLS-I	0.81 (4.65)	-0.0017 (-1.27)	-6.4×10^{-6} (-0.91)	-	0.04	[illegible]
Pokegama Lake, MN	335	ML	1.93 (8.75)	-0.0221 (-282.64)	-5.3×10^{-5} (9.04)	8.64	-	-
		OLS-I	1.44 (7.28)	-0.0033 (-4.46)	-1.4×10^{-5} (-1.57)	-	0.24	[illegible]
Pomona Lake, KS	336	ML	1.71 (1.10)	-0.0368 (-0.63)	2.4×10^{-5} (0.36)	-16.39	-	
		OLS-I	1.54 (5.35)	-0.0058 (-1.11)	8.4×10^{-6} (0.62)	-	[illegible]	

Table 6-1 (continued)

Site name	Site No.	Estimator	Intercept	T+M cost	Income	Function value	R^2	df
Proctor Lake, TN	337	ML	4.09	-0.0643	5.0×10^{-6}	-6.63	-	-
			(6.59)	(-2.14)	(0.27)			
		OLS-I	2.06	-0.0134	1.2×10^{-6}	-	0.54	49
			(13.61)	(-7.50)	(0.19)			
Rathbun Reservoir, IO	338	ML	-39.594	-0.4109	7.1×10^{-4}	-3.03	-	-
			(-35.16)	(-24.10)	(39.79)			
		OLS-I	0.77	-0.0015	1.4×10^{-5}	-	0 02	28
			(1.85)	(-0.27)	(0 82)			
Sam Rayburn Dam & Reservoir, TX	339	ML	1.60	-0.0744	1.0×10^{-5}	-14.41	-	-
			(1.64)	(-2.52)	(0.23)			
		OLS-I	1.46	-0.0094	1.0×10^{-6}	-	0.11	64
			(7.06)	(-2.83)	(0.13)			
Sardis Lake, MS	340	ML	2.48	-0.0095	1.5×10^{-5}	-100.97	-	-
			(7.01)	(-2.05)	(0.64)			
		OLS-I	1.81	-0.0030	4.3×10^{-6}	-	0.05	202
			(20.73)	(-3.17)	(0.78)			
Waco Lake, TX	343	ML	3.54	-0.0345	-1.4×10^{-4}	-27.67	-	-
			(0.32)	(-0.14)	(-0.20)			
		OLS-I	1.95	-0.0006	-7.4×10^{-6}	-	0.03	58
			(15.04)	(-0.32)	(-1.25)			
Whitney Lake, TX	344	ML	-0.378	-0.0166	3.0×10^{-5}	-98.95	-	-
			(-0.17)	(-1.04)	(0.83)			
		OLS-I	1.41	-0.0025	3.2×10^{-6}	-	0.02	201
			(13.07)	(-1.80)	(0.72)			
Youghiogheny River Lake, PA	345	ML	-37.27	0.4256	5.4×10^{4}	-11.59	-	
			(-61.88)	(25.12)	(46.56)			
		OLS-I	0.29	0.0263	1.7×10^{-5}	-	0.14	28
			(0.60)	(1.61)	(1.55)			

and McGivney [1983] and their estimated t-ratios for the null hypothesis of no association.

There are several interesting aspects of these results in comparison with the original OLS estimates. First, when travel cost was judged to be a significant determinant of site demand based on the OLS estimates, the ML estimates were generally consistent. Second, income appears more frequently to be a significant determinant of demand, though the parameter estimate is quite small. Finally, and most relevant to the use of the model for benefits analysis, the estimated parameter for the travel cost using the ML estimator is generally larger (in absolute magnitude) than the OLS estimates. Although there are a few exceptions, this pattern is fairly consistent across the sites considered.

Before the implications of these differences and the performance of the generalized travel cost model's second stage equations are considered, several caveats should be noted. One reason for differences in the estimated parameters for the travel cost variable is likely to be the result of the treatment of the value of the censoring point with the OLS estimates.* The censoring point was coded as a larger value (eight visits instead of six) in an effort to split the implicit open-ended interval. The rationale for the approximation was purely heuristic based on a common practice for interval data that uses the midpoint of the reported range. The method assumes a symmetric distribution of responses within the range of each interval. Without information on the actual maximum number of visits to each site it was impossible to determine a

*The point of censoring with the Federal Estate survey was six trips including the trip in which the respondent was interviewed. In fact, the data were actually reported in a grouped format. After this research was completed, we became aware of yet another approach using a maximum likelihood model of interval data proposed by Stewart [1983]. We are currently conducting research to compare this ML estimator with the one used in our analysis.

plausible endpoint for the open-ended class. However, based on the pattern of usage at given travel costs to each site, eight visits were judged to be a reasonable approximation.

A second issue arises with the convergence criteria for the numerical solution algorithm used to optimize the log-likelihood function. In all cases, the estimates satisfied the criteria for an optimum.* However, for several of the sites, the first partial derivatives of the log-likelihood function remained quite large at the solution point. Ordinarily these are expected to be approximately zero at the optimum.

A number of variations on the search process and convergence criteria were evaluated to reduce the size of these derivatives: (1) increasing the stringency of the convergence criteria; and (2) shifting the convergence criteria to focus exclusively on the first derivatives. Neither of these modifications affected the model estimates or markedly affected the size of these first partial derivatives. With increased stringency in the overall criteria, the parameter estimates remained approximately constant with rather small changes in the derivatives. Exclusive reliance on the size of the derivatives generally stopped the search process at whatever maximum iteration level was set without the results satisfying this convergence criteria.†

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*GQOPT allows for three convergence criterion to be used: (1) when attempted step sizes of the parameters to be estimated are each less than the prespecified accuracy level (.00001), (2) when the norm of the gradient is less than the prespecified accuracy level, and (3) when the relative improvement in the function value is less than the prespecified accuracy level. Our models uniformly converged under the first criterion.

†Olsen [1982] also notes problems with ML estimation with truncated normal errors and sample size less than several hundred observations.

It is reasonable to expect that an ML estimator for a truncated-censored model will be sensitive to the number of observations clustered at the two end-points (especially at the point of censoring) of the observed values of the dependent variable. The greater this clustering, the less the estimator will be able to discriminate responses to the independent variables. The more sophisticated statistical approach becomes ineffective when the data are clumped at the truncation or censoring points. The OLS estimates are least reliable under the same set of circumstances. The empirical implications of these features are considered in the next section.

Before continuing with the second stage of the generalized travel cost model using the ML estimates, some preliminary perspective of the implied differences in benefits is presented. To provide this perspective, the consumer surplus given by the negative of the reciprocal of the travel cost coefficient is calculated. It is derived from the indefinite integral for the semi-log demand function. Although there is no price at which the quantity demanded of use will be zero (i.e., horizontal intercept) with the semi-log specification, it is straightforward to show the basis for the approximation of the Marshallian consumer surplus. (This measure is discussed in more detail in Chapter 7.) Equation (6.7) defines the indefinite integral using a semi-log specification:*

$$A = \int e^{\alpha_0 - \alpha_1 TC} dTC = -\frac{1}{\alpha_1}(e^{\alpha_0 - \alpha_1 TC}) . \tag{6.7}$$

Since the term in parentheses at the far right of Equation (6.7) is the predicted quantity demanded, ($\frac{1}{\alpha_1}$) is an approximate measure of the consumer

*Note in this case α_1 has been explicitly identified as having a negative effect on the quantity demanded. In other sections of this report we have not explicitly identified the sign of α_1.

surplus per day. In Equation (6.7), A defines the function to be evaluated in measuring the Marshallian consumer surplus. Thus, even though the horizontal intercept for the demand function is not finite, assuming there is a choke price, TC*, then the Marshallian consumer surplus realized at price TC is given as

$$\dot{C}S = -\frac{1}{\alpha_1} e^{\alpha_0 - \alpha_1 TC^*} - \left(-\frac{1}{\alpha_1} e^{\alpha_0 - \alpha_1 TC} \right) . \qquad (6.8)$$

If TC* becomes arbitrarily large, then the first term in Equation (6.8) approaches zero and the second term is the Marshallian consumer surplus. Evaluating quantity demanded at the existing price yields $\frac{1}{\alpha_1}$ as the consumer surplus per unit of use.

Table 6-2 presents the consumer surplus per season for each of the 22 sites in the original sample in Desvousges, Smith, and McGivney [1983]. These results suggest rather large differences in the implied valuation of a visit to each site. The estimates based on the OLS estimates range from 39.00 to 400.00; those based on the ML estimates are substantially smaller, 3.58 to 105.26 (in 1977 dollars). Indeed, one of the criticisms of the earlier site demand estimates was based on this evaluation. Values of a visit to the site Were judged to be “too high” relative to the range of values usually considered relevant for these water-based recreation sites based on grouped data.† Consequently, the ML estimates would clearly be judged as more plausible based on this criterion.

†See Loomis and Sorg [1982] for a review of these estimates in a form designed to attempt to develop consistent estimates of the value per day of different types of recreation. See also Dwyer, Kelly, and Bowes [1977] for a review of the travel cost estimates.

Table 6-2. Consumer Surplus Estimates: OLS Versus ML

Site name	Site No.	Consumer surplus per season OLS	Consumer surplus per season ML
Arkabutla Lake, MS	301	107.5269	21.1416
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	80.0000	80.0000
Belton Lake, TX	304	39.6825	13.7552
Benbrook Lake, TX	305	185.1852	21.1864
Blakely Mt. Dam, Lake Ouachita, AR	307	126.5823	26.7380
Canton Lake, OK	308	48.5437	3.5868
Cordell Hull Dam and Reservoir, TN	310	71.9424	15.2207
DeGray Lake, AR	311	142.8571	37.4532
Grapevine Lake, TX	314	136.9863	32.1543
Greers Ferry Lake, AR	315	153.8462	34.8432
Grenada Lake, MS	316	105.2632	10.8225
Hords Creek Lake, TX	317	200.0000	19.9203
Melvern Lake, KS	322	126.5823	5.5648
Millwood Lake, AR	323	123.4568	30.2115
Mississippi River Pool No. 6, MN	325	135.1351	17.6991
New Savannah Bluff Lock & Dam, GA	329	149.2537	18.5874
Ozark Lake, AR	331	217.3913	43.4783
Philpott Lake, VA	333	114.9425	29.8507
Proctor Lake, TN	337	76.6269	15.5521
Sam Rayburn Dam & Reservoir, TX	339	106.3830	13.4409
Sardis Lake, MS	340	333.3333	105.2632
Whitney Lake, TX	344	400.0000	60.2410

6.4 ESTIMATING THE MODEL'S SECOND STAGE WITH NEW DATA

After the first stage of the two-stage generalized travel cost model was estimated to develop new demand functions, the second stage of the model was also estimated using the statistical modifications designed to accommodate the censored and truncated visit data. The following sections summarize the re-estimation of the model's second stage, report the new values derived with the reestimated model, and describe the revisions to the model to allow both the addition of new data on four key variables--site substitution potential, potential effect of congestion, types of fishing possible, and several alternatives measures of water quality--and the introduction of variables to account for the mix of recreation activities at a site.

6.4.1 A Comparison of OLS and ML Values

In the first part of the extended analysis, the original model using the ML estimates as data is reestimated.* The analysis is provided for several samples of sites, which differ depending on the number of Corps of Engineers sites included. For each of the samples, the summary equations for the three parameters (intercept, travel cost, and income) of the generalized travel cost model are presented in Tables 6-3 through 6-5. The first two samples of sites shown in columns 1 and 2 of each table correspond to the samples in Desvousges, Smith, and McGivney [1983]. The 22 sites were those sites with the less severe truncation effects based on the Olsen [1980] diagnostic index. The 33-site sample consists of all sites with valid data at the time of that report.

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*The ML estimates of the site demand parameters and the estimates of the asymptotic variances for them can be used as data following the same overall logic outlined in Desvousges, Smith, and McGivney [1983]. However, the Aitken generalized least-squares (GLS) estimator in this case no longer corresponds to the approach discussed by Saxonhouse [1977], since the assumed error structure and corresponding first-stage estimator are different. Nonetheless, the GLS approach still provides a way to reflect the relative precision in the estimation of each of these site demand functions.

Table 6-3. A Comparison of the GLS Estimates for the Original Model--OLS Versus ML: Intercept (α_0)

Independent variables	OLS		ML		
	22 sites	33 sites	22 sites	33 sites	42 sites
Intercept	1.5106 (4.081)	1.2959 (3.768)	-.0443 (-0.024)	-.9999 (-0.167)	-6.5599 (-2.892)
SHORMILE	0.0003 (1.250)	-0.0003 (-1.304)	.0006 (0.782)	.0019 (0.714)	.0114 (5.393)
MULTI+ACC	-0.0059 (-1.502)	0.0017 (0.464)	-.0388 (-1.071)	.0077 (0.085)	-.0015 (-0.023)
AREAP/AREAT	-0.3950 (-1.752)	-0.1686 (-1.116)	1.4607 (1.030)	.0491 (0.013)	-4.6724 (-5.914)
Mean dissolved oxygen	0.0045 (1.065)	0.0049 (1.220)	.0195 (2.076)	.0216 (0.781)	.0478 (3.153)
Variance in dissolved oxygen	0.0005 (1.862)	0.0003 (1.131)	-6.47×10^{-5} (-2.077)	-7.39×10^{-5} (-0.738)	-1.14×10^{-4} (-1.827)

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

AREAP/AREAT = Pool surface acreage on fee and easement lands during peak visitation period divided by total site area, land, and water in acres.

Table 6-4. A Comparison of the GLS Estimates for the Original Model--OLS Versus ML: Travel Cost Parameter (α_1)

Independent variables	OLS 22 sites	OLS 33 sites	ML 22 sites	ML 33 sites	ML 42 sites
Intercept	-.0246 (-9.480)	.0005 (0.203)	-.0215 (-0.431)	-.0170 (-0.329)	-.0835 (-2.880)
SHORMILE	$-.13 \times 10^{-4}$ (-6.763)	$.47 \times 10^{-6}$ (0.256)	$-.11 \times 10^{-5}$ (-0.382)	-4.38×10^{-6} (-0.141)	-1.87×10^{-5} (-0.639)
MULTI+ACC	77×10^{-4} (2.810)	$-.41 \times 10^{-4}$ (-1.586)	2.68×10^{-3} (1.301)	2.32×10^{-3} (1.357)	5.57×10^{-4} (1.452)
AREAP/AREAT	.0033 (2.273)	-.0025 (-2.190)	-.0894 (-1.522)	-.0975 (-1.944)	-.0373 (-3.828)
Mean dissolved oxygen	.0002 (5.992)	-4.2×10^{-4} (-1.514)	-.0001 (-0.286)	-9.15×10^{-5} (-0.263)	5.32×10^{-4} (2.487)
Variance in dissolved oxygen	$.98 \times 10^{-5}$ (4.077)	$-.17 \times 10^{-5}$ (-0.751)	1.48×10^{-7} (0.127)	2.63×10^{-7} (0.220)	1.67×10^{-6} (-2.296)

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

AREAP/AREAT = Pool surface acreage on fee and easement lands during peak visitation period divided by total site area, land, and water in acres.

Table 6-5. A Comparison of the GLS Estimates for the Original Model--OLS Versus ML: Income Parameter (α_2)

Independent variables	OLS 22 sites	OLS 33 sites	ML 22 sites	ML 33 sites	ML 42 sites
Intercept	$.54 \times 10^{-5}$ (0.308)	$.53 \times 10^{-5}$ (0.330)	$.17 \times 10^{-4}$ (0.657)	$.22 \times 10^{-3}$ (3.513)	1.99×10^{-4} (5.169)
SHORMILE	$.97 \times 10^{-9}$ (0.089)	$-.14 \times 10^{-7}$ (-1.408)	-6.036×10^{-8} (-1.449)	5.97×10^{-8} (0.412)	1.77×10^{-7} (1.351)
MULTI+ACC	$.47 \times 10^{-6}$ (2.562)	$.22 \times 10^{-6}$ (1.299)	1.33×10^{-7} (-0.074)	6.83×10^{-6} (1.283)	2.95×10^{-6} (0.981)
AREAP/ AREAT	$-.19 \times 10^{-5}$ (-0.181)	$.10 \times 10^{-4}$ (1.423)	8.56×10^{-5} (2.731)	4.49×10^{-5} (0.403)	-9.78×10^{-5} (-1.338)
Mean dis- solved oxygen	$-.12 \times 10^{-6}$ (-0.604)	$-.12 \times 10^{-6}$ (-0.642)	-2.425×10^{-7} (-0.766)	-2.81×10^{-6} (-3.313)	-1.96×10^{-6} (-3.618)
Variance in dissolved oxygen	$.94 \times 10^{-10}$ (0.007)	$-.73 \times 10^{-8}$ (-0.617)	5.279×10^{-10} (0.573)	7.26×10^{-9} (2.779)	6.31×10^{-9} (2.976)

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

AREAP/AREAT = Pool surface acreage on fee and easement lands during peak visitation period divided by total site area, land, and water in acres.

Estimates in columns 3 through 5 are possible as a result of new data acquired, the estimation of the ML values, and more knowledge about the sites. Column 3 provides the original 22-site sample using the ML estimate. Column 4 is a “new” and different 33-site sample. It results from deleting the 9 sites with the largest average distance traveled from the 42-site sample. In effect, this sample argues that the travel cost models’ assumptions are most likely to be violated in these cases. Column 5 presents the estimates for the full sample of sites for which data are presently available--42 of the 43 possible sites.

Several conclusions emerge from a comparison of the estimated demand parameter equations across each of the potential sources for model variation--specifically, the base estimator used (OLS versus ML) and the sample composition. With both base estimators the results appear to be sensitive to sample composition. This is unfortunate in the case of the ML estimates, because there is not as clear a rationale for accepting a particular sample. If it is maintained that the ML method adequately deals with the truncation and censoring problems, then there is no clear reason for preferring the 22-site sample over the other two samples.

Comparison of the results based on the 33- and 42-site samples for the water pollution variables (i.e., mean dissolved oxygen and variance in dissolved oxygen) suggests the parameters are quite disparate between these two samples. However, they are no more divergent than those observed with the original OLS models. Statistically significant and theoretically plausible effects for the mean dissolved oxygen variable with both the intercept and travel cost parameters using the full sample are present. Reducing the sample to 33 sites yields an insignificant estimated effect for both coefficients and a reversed sign for the coefficient associated with the model for the travel cost parameters. Surprisingly, there is greater consistency (though not necessarily more plausibility as discussed below) for the estimates based on the estimated parameter for the income. The record with the 22-site sample is consistent with the 42-site sample for the water pollution variables in the equation for the intercept. By contrast, there is considerable variation in sign and statistical significance of the estimated effects of the site characteristics for the other parameters (i.e., the travel cost and income coefficients) using this sample versus the 42-site sample.

These diverse results across samples and estimators imply that the evaluation of the benefits of a water quality improvement may be quite sensitive to the specific model selected. As noted earlier, there is no clear basis for choosing one of the samples over another. Of course, under the assumption that a truncated-censored normal error adequately describes the stochastic component of individual recreationists' demands for these sites, then the models based on ML estimates would be preferred to those derived from OLS estimates. To gauge the effects of the model selected on the benefit estimates, the analysis of water quality changes presented in Chapter 7 will use the original Desvousges, Smith, and McGivney [1983] specification for the demand parameter models. However, benefit estimates will be prepared using models that vary according to the base estimator and the sample size to identify the sources of variation in benefit estimates.

6.4.2 Estimating the Model's Second Stage: The Addition of New Data

Estimating the model's second stage requires addressing the use of the varying parameter travel cost model to reflect different activities undertaken at a water-based recreational facility. At the outset, it is important to caution that the data limitations are severe with only crude measures of the level of participation in each activity by the representative user. Although it is impossible to offset the limited data on activities, the information collected on managers' perceptions of the attributes of their respective sites to individuals' demands for these sites' services does enhance our capabilities. These additional variables are listed in Table 6-6. Although these variables are all based on a manager's judgment, they may provide a basis for judging, albeit in a limited way, the implications of some of the most restrictive assumptions required by the earlier application of the varying parameter model to the Federal Estate data.

A large number of alternative specifications were considered in developing revised estimates for the second stage of the generalized travel cost model. Specifications included all of the newly acquired variables along with other water quality measures and the indexes of participation in specific recreational activities. However, the selection of "final" models was complex, involving substantial pretesting and judgment. The results presented in Tables 6-7 through 6-15 are a small sample of of the alternative model specifications and samples used to estimate these second-stage functions. The tables report these estimates for various sample compositions and model specifications. The first three tables use the full sample of sites, and the remainder present comparable results for each of the two subsamples. The first model in each table corresponds to the original model from Desvousges, Smith, and McGivney [1983]. All of the variables in Table 6-6 were evaluated as potential determinants of the site demand parameters but the tables report only the cases that indicated a potentially important relationship. In addition, the indexes of participation in specific activities (i.e., BOAT for the index for boating and FISH for fishing) were entered as separate variables and also interacted with other variables

Table 6-6. Additional Site Characteristics Considered in Second-Stage Analysis

Variable name	Description
COLD	Qualitative variable indicating presence of coldwater gamefish (=1)
STOCK	Qualitative variable indicating the presence of an ongoing fish-stocking program (=1)
CGP	Qualitative variable indicating extreme congestion during maximum usage time periods (=1)
CGW	Qualitative variable indicating congestion on weekends with good weather (=1)
CGD	Qualitative variable indicating congestion on weekdays (=1)
CGV	Qualitative variable indicating congestion varies within a particular facility (=1); i.e., greatly varying levels of congestion within the area
SUB	Manager's judgment that no good substitutes are within the immediate area of the site (=1)
LICEN	Qualitative variable indicating additional requirement for a license to fish at a particular facility

Table 6-7. Second-Stage Models: ML Estimates and 42-Site Sample for Intercept (α_0)

Independent variables	Models 1	2	3	4	5	6
Intercept	-6.5599	-5.8201	-1.5231	-3.9496	-6.5692	4.0168
	(-2.892)	(-5.528)	(-.605)	(-1.229)	(-4.275)	(2.937)
SHORMILE	.0114	.0066	.0078	.0069	.0070	.0101
	(5.393)	(3.716)	(4.400)	(3.667)	(3.724)	(5.575)
MULTI+ACC	-.0015	.1259	.0915	.1111	.0978	.0014
	(-.023)	(2.658)	(7.550)	(1.704)	(1.541)	(.031)
ARSIZE	-4.6724	2.6666	.0398	2.9809	2.7990	-6.9980
	(-5.914)	(1.140)	(.016)	(1.252)	(1.182)	(-6.107)
DOM	.0478	.0507	.0424	.0506	.0484	.0166
	(3.153)	(4.996)	(4.106)	(4.571)	(4.486)	(3.785)
DOV	-1.14×10^{-4}	--	--	--	--	--
	(-1.827)	--	--	--	--	--
DOM*BOAT	--	-.0797	-.0547	-.0829	-.0859	--
	--	(-5.710)	(-2.703)	(-4.820)	(-5.108)	--
DOM*FISH	--	-.0137	-.0173	-.0056	.0082	--
	--	(-.660)	(-.463)	(-.136)	(.213)	--
COLD	--	--	-2.6839	--	--	-4.5024
	--	--	(-2.431)	--	--	(-4.736)
CGP	--	--	-.4519	.2642	.5390	-1.1367
	--	--	(-.534)	(.310)	(.675)	(-3.197)
STOCK	--	--	--	-2.3285	--	--
	--	--	--	(1-.929)	--	--

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1)

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fish-stocking program (=1)

Table 6-8. Second-Stage Models: ML Estimates and 42-Site Sample for Travel Cost Parameter (α_1)

Independent variables	Models 1	2	3	4	5	6
Intercept	-.0835 (-2.880)	-.0328 (-2.789)	-.0463 (-2.477)	-.0655 (-3.281)	-.0416 (-2.607)	-.0372 (-2.658)
SHORMILE	-1.87×10^{-5} (-.639)	-8.08×10^{-6} (-.296)	-1.76×10^{-5} (-.576)	-8.58×10^{-6} (-.320)	-1.17×10^{-5} (-.422)	-2.98×10^{-5} (-1.095)
MULTI+ACC	5.57×10^{-4} (1.452)	-7.44×10^{-4} (-1.299)	-7.24×10^{-4} (-1.005)	-7.79×10^{-4} (-1.318)	-9.09×10^{-4} (-1.490)	-.0011 (-1.613)
ARSIZE	-.0373 (-3.828)	.0161 (1.000)	.0290 (1.247)	.0339 (1.527)	.0294 (1.281)	.0124 (.802)
DOM	5.32×10^{-4} (2.487)	2.63×10^{-4} (2.815)	2.74×10^{-4} (2.852)	3.08×10^{-4} (3.311)	2.67×10^{-4} (2.843)	1.10×10^{-4} (1.542)
DOV	-1.67×10^{-6} (-2.296)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	-1.59×10^{-5} (-.161)	-8.67×10^{-5} (-.691)	-8.99×10^{-5} (-.780)	-6.97×10^{-5} (-.584)	-- --
DOM*FISH	-- --	-4.53×10^{-4} (-3.509)	-4.02×10^{-4} (-2.541)	-5.32×10^{-4} (-3.913)	-3.87×10^{-4} (-2.522)	-- --
COLD	-- --	-- --	.0051 (.500)	-- --	-- --	-.0017 (-.162)
CGP	-- --	-- --	.0048 (.670)	.0025 (.369)	.0056 (.819)	.0128 (2.431)
STOCK	-- --	-- --	-- --	.0262 (1.872)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-9. Second-Stage Models: ML Estimates and 42-Site Sample for Income Parameter (α_2)

Independent variables	Models 1	2	3	4	5	6
Intercept	2.00×10^{-4} (5.169)	8.93×10^{-5} (1.052)	-6.36×10^{-6} (-.535)	-2.53×10^{-4} (-2.386)	-9.70×10^{-7} (-.011)	-2.05×10^{-5} (-.375)
SHORMILE	1.77×10^{-7} (1.351)	7.63×10^{-8} (.486)	1.17×10^{-7} (.697)	4.42×10^{-8} (.322)	1.71×10^{-7} (1.114)	1.22×10^{-7} (.795)
MULTI+ACC	2.95×10^{-6} (.981)	7.80×10^{-6} (2.272)	2.63×10^{-6} (.656)	3.21×10^{-7} (.092)	2.51×10^{-6} (.628)	2.56×10^{-6} (.740)
ARSIZE	-9.78×10^{-5} (-1.338)	-2.94×10^{-4} (-3.179)	-1.32×10^{-4} (-1.187)	-2.07×10^{-4} (-2.145)	-1.42×10^{-4} (-1.295)	-1.42×10^{-4} (-1.650)
DOM	-1.96×10^{-6} (-3.618)	-7.08×10^{-7} (-1.347)	-3.62×10^{-7} (-.687)	-8.76×10^{-7} (-1.839)	-3.32×10^{-7} (-.635)	-2.69×10^{-7} (-.810)
DOV	6.31×10^{-9} (2.976)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	1.33×10^{-7} (.154)	-1.63×10^{-7} (-.171)	-6.28×10^{-7} (-.845)	2.24×10^{-7} (.275)	-- --
DOM*FISH	-- --	2.50×10^{-7} (.288)	5.49×10^{-7} (.477)	3.09×10^{-6} (2.741)	-8.61×10^{-8} (-.103)	-- --
COLD	-- --	-- --	4.42×10^{-5} (.803)	-- --	-- --	2.60×10^{-5} (.730)
CGP	-- --	-- --	8.50×10^{-5} (2.153)	1.54×10^{-4} (4.319)	6.26×10^{-5} (2.257)	7.75×10^{-5} (2.256)
STOCK	-- --	-- --	-- --	2.04×10^{-4} (3.450)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DIM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-10. Second-Stage Models: ML Estimates and 33-Site Sample for Intercept (α_0)

Independent variables	Models 1	2	3	4	5	6
Intercept	-.9999 (-.167)	-3.5596 (-.877)	-4.6396 (-.989)	-1.7705 (-.370)	-3.4641 (-.799)	1.6285 (.756)
SHORMILE	.0019 (.714)	.0041 (1.296)	.0027 (.721)	.0046 (1.376)	.0040 (1.231)	.0012 (.331)
MULTI+ACC	.0077 (.085)	-.0542 (-.616)	-.0735 (-.625)	-.0027 (-.022)	-.0493 (-.443)	.0141 (.151)
ARSIZE	.0491 (.013)	5.1222 (1.236)	6.2297 (1.350)	7.9166 (1.467)	5.1886 (1.200)	.7284 (.203)
DOM	.0216 (.781)	.0719 (1.884)	.0749 (1.738)	.0861 (1.866)	.0707 (1.675)	-.0013 (0.130)
DOV	-7.39×10^{-5} (-.738)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	-.0781 (-2.031)	-.0861 (-1.726)	-.0829 (-1.718)	-.0761 (-1.609)	-- --
DOM*FISH	-- --	-.0480 (-.853)	-.0455 (-.747)	-.0908 (-1.171)	-.0493 (-.821)	-- --
COLD	-- --	-- --	1.1752 (.704)	-- --	-- --	.4721 (.289)
CGP	-- --	-- --	.4090 (.247)	-1.0467 (-.569)	-.1104 (-.075)	-.9747 (-.800)
STOCK	-- --	-- --	-- --	-2.8121 (-.854)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fish-stocking program (=1).

Table 6-11. Second-Stage Models: ML Estimates and 33-Site Sample for Travel Cost Parameter (α_1)

Independent variables	Models 1	2	3	4	5	6
Intercept	-.0170	-.1209	- 1773	-.1224	-.1609	-.0268
	(-.329)	(-2.995)	(2.533)	(-2.399)	(-3.632)	(1.841)
SHORMILE	-4.38×10^{-6}	8.55×10^{-6}	3.11×10^{-5}	3.67×10^{-5}	3.14×10^{-5}	1.83×10^{-5}
	(-.141)	(.306)	(1 037)	(1.264)	(1.066)	(.560)
MULTI+ACC	.0023	.0018	.0012	.0032	.0016	.0032
	(1.357)	(1.132)	(.627)	(1.735)	(1.060)	(1.846)
ARSIZE	-.0974	.0964	.0624	.0919	.0514	-.1537
	(-1.944)	(1.154)	(.675)	(1.063)	(.616)	(-2 652)
DOM	-9.15×10^{-5}	9.42×10^{-4}	.0014	.0014	.0013	-9.67×10^{-6}
	(-.263)	(2.428)	(2 311)	(3.395)	(3.082)	(-.079)
DOV	2.63×10^{-7}	--	--	--	--	--
	(.220)	--	--	--	--	--
DOM*BOAT	--	-7.46×10^{-4}	-.0015	-.0010	-.0013	--
	--	(-2.057)	(-1.864)	(-2.917)	(-2.826)	--
DOM*FISH	--	-1.58×10^{-3}	-.0013	-.0020	-:0013	--
	--	(2.814)	(-2.168)	(-2.702)	(-2.308)	--
COLD	--	--	.0071	--	--	.0179
	--	--	(.307)	--	--	(-1.274)
CGP	--	--	.0444	.0266	.0407	.0289
	--	--	(1.727)	(1.114)	(1.829)	(1.686)
STOCK	--	--	--	-.0562	--	--
	--	--	--	(-1.430)	--	--

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-12. Second-Stage Models: ML Estimates and 33-Site Sample for Income Parameter (α_2)

Independent variables	Models 1	2	3	4	5	6
Intercept	2.16×10^{-4} (3.513)	3.00×10^{-4} (1.639)	5.02×10^{-4} (2.381)	2.16×10^{-4} (1.241)	4.98×10^{-4} (2.367)	2.56×10^{-5} (.343)
SHORMILE	5.97×10^{-8} .(.412)	-4.31×10^{-8} (-.235)	-2.91×10^{-7} (-1.270)	-2.79×10^{-7} (-1.872)	-1.75×10^{-7} (-.908)	-1.05×10^{-7} (-.436)
MULTI+ACC	6.83×10^{-6} (1.283)	1.33×10^{-5} (2.457)	1.43×10^{-5} (2.693)	3.27×10^{-6} (.669)	1.48×10^{-5} (2.801)	9.36×10^{-6} (1 559)
ARSIZE	4.49×10^{-5} (.403)	-2.81×10^{-4} (-1.572)	-4.24×10^{-4} (-2.264)	-5.75×10^{-4} (-3.903)	-3.97×10^{-4} (-2 151)	-9.24×10^{-5} (-.732)
DOM	-2.81×10^{-6} (-3.313)	-4.78×10^{-6} (-3.514)	-6.75×10^{-6} (-3.977)	-6.43×10^{-6} (-4.989)	-6.61×10^{-6} (-3.918)	-9.98×10^{-7} (-1.688)
DOV	7.26×10^{-9} (2.779)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	5.06×10^{-6} (2.901)	7.36×10^{-6} (3.447)	6.34×10^{-6} (3.855)	7.32×10^{-6} (3.434)	-- --
DOM*FISH	-- --	9.56×10^{-7} (.498)	2.05×10^{-6} (1.060)	5.87×10^{-6} (3.352)	1.88×10^{-6} (.979)	-- --
COLD	-- --	-- --	7.26×10^{-5} (.941)	-- --	-- --	5.95×10^{-5} (.643)
CGP	-- --	-- --	-7.89×10^{-5} (-1.302)	2.27×10^{-5} (.434)	-9.80×10^{-5} (-1.718)	5.39×10^{-5} (.935)
STOCK	-- --	-- --	-- --	3.22×10^{-4} (4.176)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-13. Second-Stage Models: ML Estimates and 22-Site Sample for Intercept (α_0)

Independent variables	Models					
	1	2	3	4	5	6
Intercept	-.0042	1.0886	2897	1.7509	1.1207	2.8109
	(-.024)	(.690)	(168)	(.807)	(.685)	(3.799)
SHORMILE	5.94×10^{-4}	1.34×10^{-3}	$5\ 00 \times 10^{-4}$	1.30×10^{-3}	1.31×10^{-3}	-6.00×10^{-4}
	(.782)	(1.161)	(.378)	(1.060)	(1.095)	(-.517)
MULTI+ACC	-.0388	-.0759	- 0637	-.0574	-.0699	-.0072
	(-1.071)	(-1.591)	(-1 175)	(-.911)	(-1.263)	(-.176)
ARSIZE	1.4607	1.5899	2 5073	1.8609	1.7791	.5191
	(1.030)	(.865)	(1 201)	(.875)	(.864)	(355)
DOM	.0194	.0181	.0215	.0190	.0173	-.0018
	(2.076)	(1.107)	(1.256)	(1.053)	(1.009)	(-.472)
DOV	-6.47×10^{-5}	--	--	--	--	--
	(-2.077)	--	--	--	--	--
DOM*BOAT	--	-.0232	-.0289	-.0224	-.0217	--
	--	(-1.458)	(-1 596)	(-1.226)	(-1.229)	--
DOM*FISH	--	3.11×10^{-3}	$-2\ 88 \times 10^{-3}$	-4.77×10^{-3}	9.65×10^{-4}	--
	--	(.172)	(-.141)	(-.193)	(.047)	--
COLD	--	--	.7230	--	--	.4617
	--	--	(1.299)	--	--	(.853)
CGP	--	--	.0965	-.2078	-.1320	-.3992
	--	--	(.169)	(-.348)	(-.237)	(-.921)
STOCK	--	--	--	-.6713	--	--
	--	--	--	(-.461)	--	--

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-14. Second-Stage Models: ML Estimates and 22-Site Sample for Travel Cost Parameter (α_1)

Independent variables	Models					
	1	2	3	4	5	6
Intercept	-.0215 (-.431)	-.1272 (-3 181)	-.0701 (-.760)	-.0429 (- 378)	-.1451 (-3.341)	- 0240 (-2.100)
SHORMILE	-1.07×10^{-5} (-.382)	1.47×10^{-5} (.606)	2.71×10^{-5} (914)	3.44×10^{-5} (1.175)	3.14×10^{-5} (1.079)	2.89×10^{-5} (1.002)
MULTI+ACC	.0027 (1.301)	.0037 (2.107)	.0043 (1.770)	0034 (1.716)	.0029 (1.534)	0031 (1 747)
ARSIZE	-.0894 (-1.522)	.0756 (.993)	-.0302 (-.273)	.0322 (.373)	0334 (.388)	- 1706 (-2.918)
DOM	-1.02×10^{-4} (-.286)	9.20×10^{-4} (2.402)	4.54×10^{-4} (.556)	1.14×10^{-3} (2.705)	1.10×10^{-3} (2.623)	1.33×10^{-5} (.112)
DOV	1.48×10^{-7} (.127)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	-7.35×10^{-4} (-2.161)	-1.91×10^{-4} (-.184)	-1.10×10^{-3} (-2.384)	-1.05×10^{-3} (-2.306)	-- --
DOM*FISH	-- --	-.0018 (-3.018)	- 0014 (-1.957)	-.0016 (-2.283)	-.0015 (-2.157)	-- --
COLD	-- --	-- --	-.0250 (-.922)	-- --	-- --	- 0276 (-2.178)
CGP	-- --	-- --	.0141 (.448)	.0292 (1.070)	.0284 (1.041)	.0411 (2.404)
STOCK	-- --	-- --	-- --	-.1081 (-.971)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

Table 6-15. Second-Stage Models: ML Estimates and 22-Site Sample for Income Parameter (α_2)

Independent variables	Models					
	1	2	3	4	5	6
Intercept	1.69×10^{-5} (.657)	-3.24×10^{-5} (-.588)	-7.51×10^{-5} (-1.762)	-1.35×10^{-4} (-1.993)	-7.29×10^{-5} (-.857)	5.98×10^{-6} (.575)
SHORMILE	-6.04×10^{-8} (-1.449)	-3.84×10^{-8} (-.762)	-1.44×10^{-7} (-3.658)	-3.45×10^{-8} (-.667)	-1.10×10^{-8} (-.163)	-1.91×10^{-7} (-5.162)
MULTI+ACC	1.35×10^{-7} (.074)	-3.35×10^{-8} (-.016)	1.47×10^{-7} (.112)	-4.09×10^{-6} (-1.853)	-9.39×10^{-7} (-.361)	1.20×10^{-6} (1.065)
ARSIZE	8.56×10^{-5} (2.731)	1.13×10^{-4} (2.017)	1.03×10^{-4} (2.890)	4.35×10^{-5} (.715)	1.39×10^{-4} (1.983)	3.97×10^{-5} (1.877)
DOM	-2.43×10^{-7} (-.766)	-2.67×10^{-7} (-.561)	-7.35×10^{-8} (-.177)	-5.73×10^{-7} (-.856)	1.57×10^{-7} (.190)	-2.55×10^{-7} (-2.513)
DOV	5.28×10^{-10} (.573)	-- --	-- --	-- --	-- --	-- --
DOM*BOAT	-- --	-2.08×10^{-8} (-.036)	-5.62×10^{-7} (-1.141)	-8.19×10^{-8} (-.107)	-5.17×10^{-7} (-.526)	-- --
DOM*FISH	-- --	5.23×10^{-7} (1.077)	6.53×10^{-7} (2.461)	2.34×10^{-6} (3.312)	4.14×10^{-7} (.789)	-- --
COLD	-- --	-- --	7.47×10^{-5} (6.541)	-- --	-- --	7.04×10^{-5} (5.244)
CGP	-- --	-- --	3.56×10^{-5} (3.189)	4.08×10^{-5} (2.240)	1.35×10^{-5} (.635)	3.12×10^{-5} (3.786)
STOCK	-- --	-- --	-- --	1.15×10^{-4} (3.310)	-- --	-- --

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fishstocking program (=1).

in the equation (e.g., water quality). The average level of dissolved oxygen (DOM) is used as the measure of water quality in all the tables. The RFF and NSF indexes also were evaluated but neither performed as well as DOM. This is most likely due to the reduced variability across sites of the indexes because of insufficient data on some variables (see Table 4-8 in Chapter 4).

Several aspects of these results should be noted. In many cases, the signs of the estimated coefficients do not agree with what would be expected a priori. For example, in the case of the intercept equations where a priori predictions are probably the most direct, the interactions between the water quality and activity index variables (e.g., DOM*BOAT and DOM*FISH) were expected to be positive. This result was not observed with any of the models with statistically significant (at conventional levels) parameter estimates for these variables. A positive effect was also expected for the qualitative variables for the presence of cold water fishing and a fish stocking program. Neither were these results observed for any of the models or samples with statistically significant parameter estimates for these variables.

However, there were also several areas of consistent results. The measure of water quality (the average dissolved oxygen level--DOM) entered individually usually had a positive and often a statistically significant effect on the intercept, indicating that improved water quality, ceteris paribus, increases the quantity demanded of a site's services. This was always the case when DOM had a statistically significant parameter estimate. Measures of the size (SHORMILE and ARSIZE) generally have positive effects on the intercept. While not always statistically significant, these results agree with a priori expectations and, in at least one case, provide an improvement over estimates of the second stage equations based on the OLS estimates for site demand parameters. Several of the site characteristics derived from the survey of site managers also have estimated effects which agree with our expectations. For example, the qualitative variable for the presence of cold water (i.e., game) fish, for some samples, has a positive but insignificant effect on the intercept.

However, it is important to avoid overinterpretation of what effects can be anticipated a priori. While intuition would seem to support other sign expectations, this is not always the case. For example, consider the case of our congestion measure. It was also expected that the presence of congestion during peak periods (i.e., with CGP = 1) would lead to a reduced demand for a site's services. However, this a priori expectation may not be correct because this effect can be difficult to judge. The congestion measure relates only to the peak periods, and it may simply indicate that individuals changed the timing of their use of sites; i.e., avoided peak periods and increased use.* Since our analysis of site demand is for the season as a whole, the demand model would be capable of reflecting such responses. Thus, despite the negative and significant estimated coefficient for CGP in one of the models for the intercept with the 42-site sample, this cannot be said to confirm a clearcut prior hypothesis.

With respect to the remaining parameters, a priori hypotheses are even more difficult to formulate. For example, consider the role of the indexes of the activities undertaken during a visit to the site. Improvements in water quality might be expected to increase the inelasticity of the site demand--it would be harder to find substitute sites, and the site would support a wider range of activities. Thus, the coefficient for the water quality variable would

*See Smith [1981] and McConnell and Sutinen [1984] for more detailed discussions for the effects of congestion on travel cost demand models.

be expected to be positive. The semi-log form implies that the travel cost parameter is a scaled version of the price elasticity.

$$\alpha_1 = \frac{1}{V}\frac{\partial V}{\partial TC} \tag{6.9}$$

where

V = quantity demanded (i.e., trips to the site)

TC = travel cost.

Thus, $\alpha_1 TC$ will be the price elasticity of demand. A positive coefficient for DOM in the second-stage equation will reduce the absolute magnitude of α_1 and therefore move the elasticity toward the inelastic region. This result was generally observed in both the original Desvousges, Smith, and McGivney [1983] analysis and in the models based on the ML estimates with statistically significant coefficients for DOM. The interaction terms of DOM with activities could reinforce this effect (i.e., also have positive coefficients) by reflecting the limited range of sites supporting these activities with the improved water quality (i.e., fewer substitutes).

However, this conclusion is somewhat ambiguous. Aggregation of demand functions across activities need not change the price elasticity of demand (or the slope with respect to price of a semi-log demand function). This result can be illustrated with a simple, two-activity example. Let $f_B(TC)$ designate the demand for a site's services resulting from participation in boating and $f_F(TC)$ the demand for the same site's services because of participation in fishing. Aggregate demand for the site's services is the sum of these demands--say $V = f_B(TC) + f_F(TC)$. The price elasticity for the aggregate demand is then

$$\varepsilon_V = \frac{\partial V}{\partial TC} \cdot \frac{TC}{V} = \left(\frac{df_B}{dTC} + \frac{df_F}{dTC} \right) \cdot \frac{TC}{V} \tag{6.10}$$

or

$$\varepsilon_V = \frac{V_B}{V} \varepsilon_B + \frac{V_F}{V} \varepsilon_F \tag{6.11}$$

where

V_j = quantity demanded of a site's services for activity j

ε_j = price elasticity of demand from the jth derived demand (j = B, F).

Thus, increases in the set of activities undertaken need not change the elasticity. It is impossible to say a priori because the outcome will depend on the underlying demand elasticities for each activity and the composition of uses of the site. Shifts in the levels of production of activities from elastic to inelastic site demands can move the elasticity of the aggregate demand toward the inelastic region. Alternatively, the opposite effect could occur from a compositional change that moves toward increased participation in the activities with more elastic demands. As a consequence, a clearcut a priori hypothesis for these terms seems unavailable. The empirical results indicate uniformly negative effects on the travel cost parameter, potentially offsetting the contribution of DOM alone to changes in the travel cost parameter, as the individual activity participation indexes approach one.

The a priori expectations for the income parameter are also limited. In general, improvements in a site's water quality should increase the income elasticity of demand. Changes in the mix of activities undertaken do not have clearcut implications for the income term. Since the estimated income parameters were often insignificantly different from zero and in 12 cases had negative estimated parameters (five of these were significantly different from zero using conventional criteria), little hope existed for the second-stage model to provide insight into the role of site attributes for a site's demand function.

However, the pessimism for the income parameter was not totally justified. Some informative results seem to have been derived. Water quality generally had a negative effect on the estimated income parameter, the opposite of a priori expectations. Nevertheless, the statistical significance of the estimated coefficient was sensitive to both the specification of the second-stage model and the composition of the sample. In general, the interaction terms (i.e., DOM*BOAT and DOM* FISH) provided mixed results. The interaction of water quality with the fishing participation index had the most theoretically consistent effects on the income parameter across models and samples, generally with positive effects that would be judged to be significantly different from zero with several specifications. This was especially true for the models based on 22 sites. With this reduced sample, the qualitative variable, stocking with cold water fish, also indicated a plausible and statistically significant effect on income. Otherwise, the findings were too mixed to suggest it was possible to observe overall tendencies in the effects of these factors on site demands.

6.5 SUMMARY

This chapter has described a set of amendments to the generalized travel cost model developed by Desvousges, Smith, and McGivney [1983]. Although all of the proposed changes were intended to improve the model's ability to take into account site characteristics in its description of recreational decisions, not all of the amendments can be judged to be successful. Improvements to the data (i.e., enhancing the number of sites that could be considered and the variables describing site characteristics) offered some new information, but did not, on the whole, enhance our understanding of the role of water quality in recreation decisions. Subjective information from site managers was of considerable interest from a descriptive perspective but did not appear to have a consistent and important influence on the second-stage models.

The ML estimator, adapted to take into account the truncated-censored nature of the dependent variable, appears reasonably successful. The parameter estimates confirm the earlier OLS results for the second-stage model in that water quality was found to have a significant and plausible effect on at least one of the demand parameters. In addition, the maximum likelihood estimates implied smaller per unit consumer surplus estimates that are more in line with other empirical findings on the value of water-based recreation. Yet even these estimates based on an improved statistical estimation procedure are not without limitations. The assumption of a normally distributed error term is an important restriction to the generality of the findings. Thus, while the maximum likelihood approach improves the ability to assess censoring and truncation effects, it has not completely solved all statistical problems with using this type of survey data.

Finally, the attempt to incorporate the role of the mix of activities undertaken on site as a determinant of observed variations in demand must be judged as incomplete. The results for the activity indexes were sensitive to both the sample composition and the specifications for the second-stage models. In many respects, the effort faced significant problems from the outset. While the theory underlying the travel cost model provides a clear role for measures of activities in the demand functions, the available data are simply not up to the tasks implied by theory. Even under the limiting assumptions employed here, information does not exist that is capable of reflecting an individual's activities as determinants of his demand for a site. Thus, the proxy variable reflected a fairly crude way to include mix of activities undertaken at a site. Clearly this was inconsistent with theory and may provide an explanation for the ambiguous findings.

CHAPTER 7

ESTIMATING THE RECREATION BENEFITS OF WATER QUALITY IMPROVEMENTS

7.1 INTRODUCTION

This chapter describes the benefit estimates from the new specifications of the generalized travel cost model. This model provides the basis for estimating the effects of water quality improvements on the demand for and valuation of a recreation site's services. As discussed in Chapter 6, the new specifications include the effects of the activities undertaken on the site on the value of a site's services, the revisions to account for the censored and truncated dependent variable, and the extensions from an augmented data base to allow for more flexibility in the estimation.

As background for using the generalized travel cost model to estimate the benefits of improved water quality, this chapter reviews the Marshallian and Hicksian measures of changes in welfare. It considers both the conceptual and practical problems that arise in using these measures to estimate the benefits of improved water quality. The chapter also reports the benefits estimates and addresses their sensitivity to the model's various specifications.* The final model estimated will depend on the specification--e.g., are activities included?--and the sample of sites providing the data--e.g., is it the full sample or a subsample?

Specifically, Section 7.2 briefly reviews the concepts of consumer surplus--both the Marshallian and the Hicksian measures and how they were used

*This argument implies that we should consider adapting our criteria for evaluating estimators to the end uses for the resulting parameter estimates. For an interesting example and discussion of the implications of this argument, see Klein et al. [1978].

to estimate the value of water quality improvements. Section 7.3 describes the selection of a “final” model and its effect on the valuation of sites. Section 7.4 discusses the difficulties in valuing water quality improvements and implications of including activity mix in those calculations. Section 7.5 summarizes the chapter and discusses its implications for benefits measurement.

7.2 BENEFIT CONCEPTS

Economists have spent the better part of this century debating the possibility of defining, and more recently estimating, dollar measures of utility changes. A central question from this debate can be expressed in: What is the change in well being an individual receives from an increase in the services of a recreation site he consumes? For this study, this question implies the need for a clear definition of its estimates of welfare changes and their relationship to the theoretically ideal counterparts.* Three measures of the benefits an individual receives from a recreation site or from improvements in one of its attributes, in this study water quality, are used: two measures of the Marshallian consumer surplus and one measure of the Hicksian compensating variation based on Hausman’s [1981] quasi-expenditure function.

Marshallian consumer surplus is probably the most well known (and often maligned) of these measures. It is the difference in the amount an individual would be willing to pay for a good with a constant per-unit price and given income and the amount actually paid. In Figure 7-1, the individual (Marshal-

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*Excellent analytical summaries of the various concepts used to measure the benefits an individual receives as a result of a price or a quantity change are available in the literature [see Freeman, 1979; Just, Hueth, and Schmitz, 1982; and McKenzie and Pearce, 1982 as examples). Moreover, Morey [1984a] has recently drawn together the literature debating the relationship between what has been defined as dollar measures of benefits and the utility changes that accompany the changes being valued.

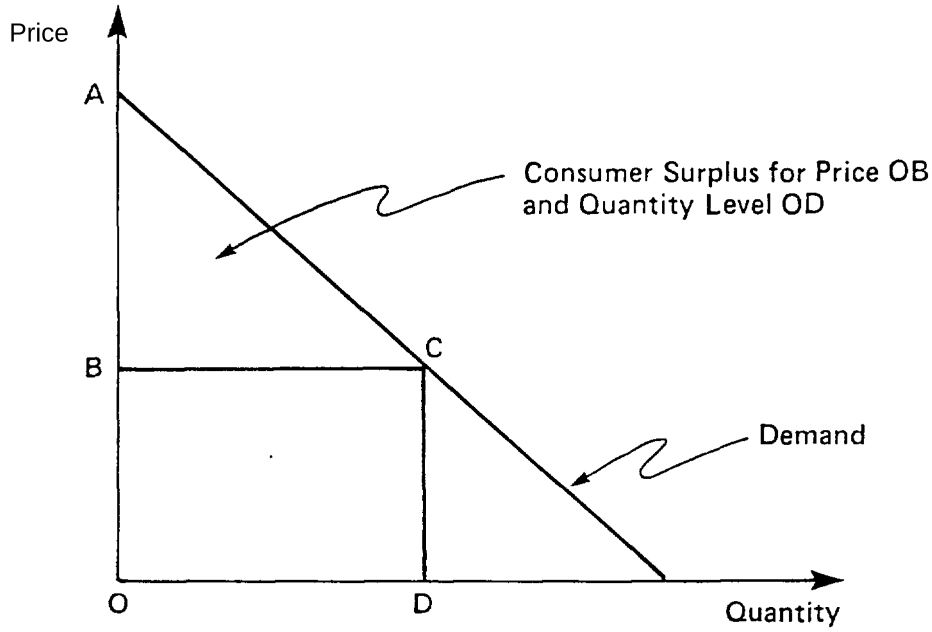

Figure 7-1. Definition of Marshallian consumer surplus.

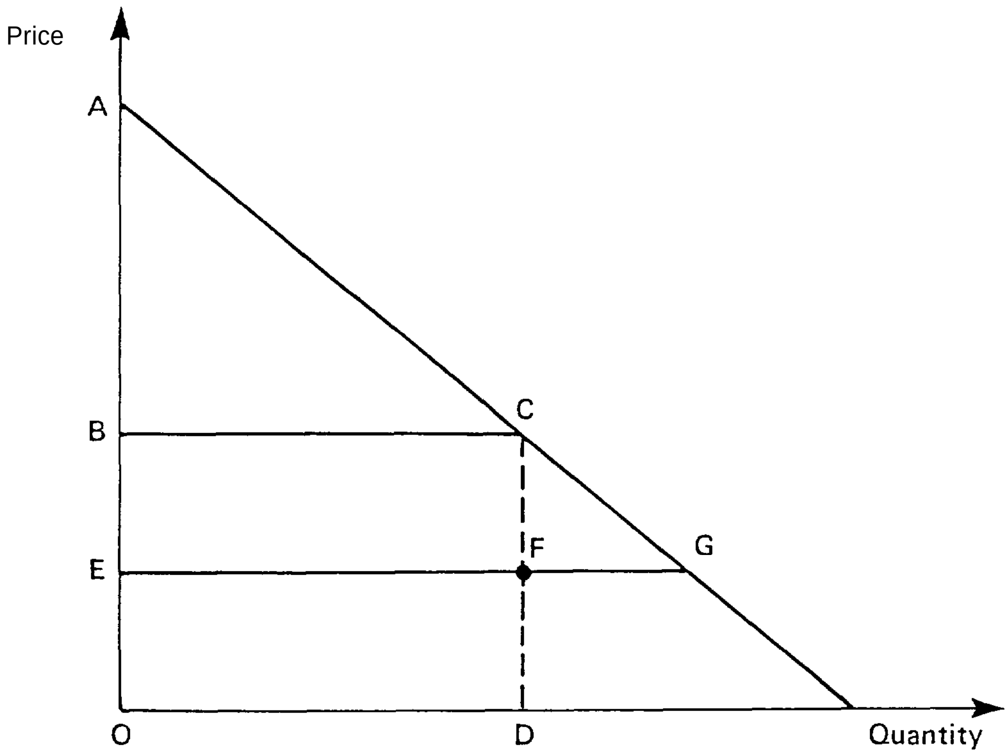

Figure 7-2. Marshallian consumer surplus for a price change.

lian) demand function describes the maximum amounts that an individual would pay for each quantity of the good for a constant level of income. The area under the demand function--the total of the maximum amounts that would be paid for each unit--is the total willingness to pay for the good. At a constant price the individual will pay less, for any level of consumption, than this maximum amount. Marshall labeled this difference between the total willingness to pay and the total expenditures as the consumer's surplus. In terms of Figure 7-1, it is the area ABC.

Until quite recently, welfare measurement has focused on valuing price changes. How much "better off" is the individual after the price of some good decreases? Indeed, nearly all of the classical discussions of consumer surplus and the original definitions of Hicks' surplus measures of welfare changes were expressed in terms of a price change. For example, the Marshallian measure of welfare change for a price change from OB to OE in Figure 7-2 is the addition (of EBCG) to the consumer surplus resulting from the price reduction. In this case, the Marshallian consumer surplus has two components: the reduced expenditures (EBCF) for the original OD units and the consumer surplus (FCG) on the additional units demanded at the reduced price.

The Marshallian consumer surplus measure has been criticized as an inadequate measure of welfare change because it does not hold an individual's utility constant. In effect, each point on a Marshallian demand curve leads to a different level of total utility. Thus, there is no clearcut relationship between the change in consumer surplus and the utility change an individual would obtain from a price change that would improve his circumstances. On the other hand, the more appropriate Hicksian welfare measure, which holds utility constant, is the largest payment he would make for the lower price set without reducing his total utility. Because the Marshallian demand function holds income and not utility constant, the individual's well-being could change as a consequence of any payments that might be made for the price change. Depending upon the size of the income effect, an individual's indifference between the payment of a consumer surplus increment with a reduced set of prices and no payment with the original prices may no longer hold true.

The Hicksian compensating variation was defined to take account of this possibility, by measuring surplus along a constant utility (rather than income) demand function. This demand function is given in Figure 7-3 along with a Marshallian demand function to illustrate the potential difference in the benefits measures for a price decrease from OB to OE. The difference (CHG) is the inaccuracy in the Marshallian measure of the change in welfare. When income effects are small, the differences between the Marshallian and Hicksian demands narrow and discrepancies in measures of the maximum willingness to pay become smaller.*

In practice, benefits analyses for environmental resources rarely, if ever, involve price changes. These analyses estimate the value of a quantity or quality changes in a resource. Unfortunately, the concern for an ideal benefit measure has, until recently, distracted attention from important practical aspects of the application of these benefits concepts in the evaluation of environmental resources. Two of the most important of these practical issues are the measurement of the demand function for the resource (or quality dimension of the resource) and the problem of translating the proposed change in the

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*The Willig [1976] bounds provide an analytical description of this narrowing for the case of price changes. Randall and Stoll [1980] have extended to the corresponding compensating and equivalent surplus concepts for quantity changes.

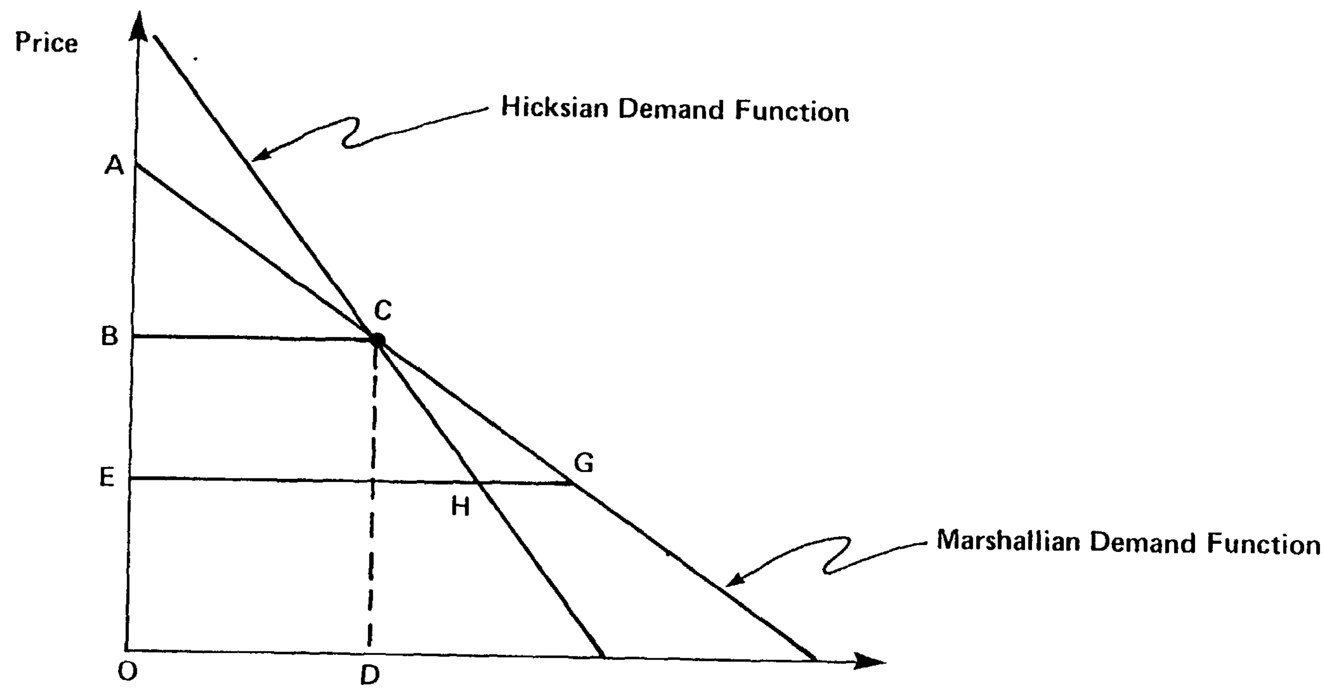

Figure 7-3. Marshallian versus Hicksian surplus measures.

resource into terms commensurate with the available demand function.* In the case of improved water quality at a recreation site, the travel cost model provides the basis for estimating the demand function and the varying parameter framework yields the information necessary to address the translation problem. However, it is important to explicitly define how the model will be used to develop estimates of the benefit measures.

The varying parameter model specifies the relationship between a site's characteristics and a "representative" individual's demand for the site's services. To value a change in one or more of the characteristics (e.g., water quality) with a Marshallian surplus measure, the model is used to predict the new demand after the change in the characteristic(s) and to estimate the change in consumer surplus. In abstract terms, this calculation is given in Equation (7.1) for the case of one characteristic, C:

$$\Delta MCS = \int_{\bar{P}}^{P(C_N)} f[p,y,\beta(C_N)]dp - \int_{\bar{P}}^{P(C_O)} f[p,y,\beta(C_O)]dp \qquad (7.1)$$

where

ΔMCS = change in Marshallian consumer surplus

$f(\cdot)$ = demand function of the services of water-based recreation sites with characteristic C_i for the representative individual

$\bar{P}$ = price (travel cost) for the representative individual

$P(C_i)$ = choke price when site characteristic is at level C_i (i.e., price when quantity demanded is zero)

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*For example, in valuing the introduction of a new recreation site, early applications of the travel cost model required one to know the demand for an existing site that was comparable (i.e., provided equivalent services) to the new site and evaluate the change as a price change to prospective users. See Cicchetti, Fisher, and Smith [1976] for an example of this approach and Morey [1984b] for an alternative approach and critique of the limitations of this earlier approximation.

$\beta(C_i)$ = model of effects of characteristic on demand parameters (i.e., second stage of the generalized travel cost model)

C_N, C_O = new (N) and initial level (0) of the characteristic that is to be evaluated.

y = income.

This approach requires that the $f(\cdot)$ and $\beta(\cdot)$ functions be correctly specified and estimated. For valuing water quality improvements, the role of site characteristics, which affect the parameters of the demand functions for site services (e.g., coefficients for the intercept and travel cost), is especially critical because the parameters from these demand equations connect changes in demand with changes in site characteristics to enable the calculation of benefits.

The compensating variation (for price changes, compensating surplus for quantity changes) is the second benefit measure used in this study. The theoretical literature implies that the Hicksian demand function or the expenditure function be known in order to use this measure of willingness to pay. In addition, the role of the site's characteristic(s) must be established in either the demand or the expenditure function. If this can be accomplished, the definition of the Hicksian measure (ΔHCS) is given as

$$\Delta HCS = E(\bar{P}, \tilde{P}, C_O, \bar{V}) - E(\bar{P}, \tilde{P}, C_N, \bar{V}) \tag{7.2}$$

where

$E(\cdot)$ = the expenditure function

$\bar{V}$ = initial utility level

$\tilde{P}$ = vector of prices for all other goods' prices.

While both Marshallian and Hicksian measures seem straightforward, implementing them can become complicated. For example, in the generalized travel cost model, a semi-log specification for the travel cost demand provided the best description of demand. In this case there is no finite “choke” price for the Marshallian measure. Rather, the demand function approaches the horizontal (price) axis asymptotically. There are two options for estimating benefits under these circumstances: (1) to use the indefinite integral and a limiting argument (as described in Chapter 6) to estimate consumer surplus per unit of use with the negative reciprocal of the parameter for the travel cost; or (2) to select an upper bound for the price, assume that it represents the choke price, and calculate the consumer surplus as in Equation (7.1).

Neither approach is ideal. The first maintains that the demand function describes behavior over the full range of prices, despite the absence of a choke price. By contrast, the second approach selects arbitrarily a choke price, facing issues similar to those discussed by Carson and Mitchell [1983] and Hanemann [1984] in their respective critiques of the Bishop-Heberlein [1979] analysis. While somewhat arbitrary, the selection of some bound provides a strategy that is more consistent with the conclusions of both of these papers’ arguments than the assumption of an unbounded choke price. Consequently, it is the primary basis for the benefit calculations in this report. However, for comparison purposes the per unit of use values implied by the negative reciprocal approach are provided for increments to the site value associated with improvements in water quality.

Practical estimation of the Hicksian measure of willingness to pay also faces problems. While it is possible to apply Hausman’s [1981] arguments with an estimated Marshallian demand function and derive the corresponding quasi-indirect utility function, this analysis has also been directed toward evaluation of price changes. Changes in one or more of the characteristics of a site raises additional implementation questions. More specifically, the Hausman analysis uses Roy's identity and the Marshallian demand function to define a partial differential equation in price(s) and income. Integration of this function yields the quasi-indirect utility function that is unique up to a monotonic transformation. Inverting the function to solve for total expenditures as a function of the utility level,* prices, and other determinants of demand, yields the quasi-expenditure function. In terms of our semi-log form for each site's demand, the differential equation is given as

$$- \frac{\frac{\partial V}{\partial TC}}{\frac{\partial V}{\partial Y}} = e^{\alpha_0 + \alpha_1 TC + \alpha_2 Y} \tag{7.3}$$

where

V(TC,Y) = quasi-indirect utility function

or

$$\frac{dY(TC)}{dTC} = e^{\alpha_0 + \alpha_1 TC + \alpha_2 Y} . \tag{7.4}$$

This can be solved for the quasi-indirect utility function given in Equation (7.5):

$$\bar{V} = -\frac{1}{\alpha_2} e^{-\alpha_2 Y} - \frac{1}{\alpha_1} e^{\alpha_0 + \alpha_1 TC} \tag{7.5}$$

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*Actually what is involved in this approach is a function of the other goods' prices and the utility level. This function can be viewed as describing the constant of integration whose value is determined by the initial conditions used in the solution to the differential equation. Discussions and correspondence with Michael Hanemann were most helpful in clarifying this relationship for the case of single versus multiple price changes.

and the quasi-expenditure function in Equation (7.6):

$$E(TC,\bar{V}) = -\frac{1}{\alpha_2}\log\left(-\alpha_2\bar{V} - \frac{\alpha_2}{\alpha_1}e^{\alpha_0+\alpha_1 TC}\right). \tag{7.6}$$

This quasi-expenditure function is used to evaluate the compensating variation measure of the benefits for a price change. The estimation requires calculating the quasi-utility level, $\bar{V}$, for the initial price and income from Equation (7.5), and then evaluating the expenditures required to obtain this level under the new price from Equation (7.6).* Given the assumptions underlying the Hausman derivation, this difference in expenditures (old minus new for a price reduction) will be the compensating variation.

Using this framework to value a quality change, rather than a price change, is not straightforward. The generalized travel cost model allows the parameters (α_0, α_1, and α_2) to be functions of site characteristics which would allow the characteristics to enter Equation (7.6) through the parameters and, in principle, evaluate the expenditures for two different levels of one or more of the site characteristics, holding prices and quasi-utility levels constant. This would seem to be the quality analog of the compensating surplus as defined in Equation (7.2).

However, there are conceptual problems with this measure. This difference (i.e., $E(TC^a, \bar{V}, \alpha(C_O)) - E(TC^a, \bar{V}, \alpha(C_N))$) evaluated using the true expenditure function would be the equivalent of McConnell's [1983] total resource value of the quality change from C_O to C_N. In the absence of weak

*Another interpretation of this process is that we are using a set of initial conditions to determine the constant of integration. Since the constant of integration will include the realized utility level, calculation of the compensating variation is possible provided this value is held fixed and the other arguments of the constant of integration function are also fixed.

complementarity, it would include the user and existence values (under McConnell's definition, it is the value of a site when use value is zero; i.e., no use is required) for the quality change. Since our analysis uses a quasi-expenditure function in which quality enters through the parameters of the demand function, existence values are also inferred. These values arise because quality-induced changes in the demand-function parameters can imply differences in both the choke prices and the expenditures to keep utility constant. These expenditures (evaluated at the choke price) would be interpreted as a measure of existence value when, in fact, they may result from features of the site demand function outside the range of variation observed in our sample. This problem would not arise under the assumption of weak complementarity, which implies that $\frac{\partial E}{\partial C_i} = 0$ for all travel costs equal to or greater than the choke price. Under this assumption, the expenditures at the different quality levels would be equal. There are no restrictions to the varying parameter model that ensure this consistency would take place because each demand parameter can respond differently to a change in the characteristic.

There are also empirical problems that are caused by the semi-log form of the demand equation and the approach used to incorporate site characteristics. Both have important implications for benefits analysis. The first problem arises because, as noted, the semi-log form of the demand function does not imply a finite choke price. Consequently, it would rely on the behavior of the quasi expenditure function to convey existence values for arbitrarily specified maximum prices. The second problem stems from the required integrability conditions. Integration requires the demand functions to be single valued, differentiable, and smooth; possess a symmetrical Slutsky-Hicks substitution matrix; satisfy the budget constraint, and, finally, have a negative semi-definite substitution matrix. This last condition is important to the semi-log form. It implies that the quantity demanded at the selected maximum price must be bounded from above by $|\alpha_1|/\alpha_2$ (see Hanemann [1979]). This condition also restricts the set of predictions for α_1 and α_2 that can be accepted for consistent benefit estimation. For our cases, it implies that negative predictions for α_2 would be inconsistent with a quasi-concave utility function. To deal with this problem, each predicted value of α_2 was checked before calculating the Hausman version of the Hicksian surplus. Negative predictions were replaced with a small positive value within the range of the estimates for the site demand function.

To avoid the first problem, the user value of an improvement in water quality is our exclusive focus. This value can be defined for our expenditure functions by choosing a fixed choke price. This solution is analogous to considering the problem to be the valuation of two sites identical in all respects except in the attribute (water quality) under study. The Hausman approach could be used in this case to derive the user value for each site, and the difference between these values would be the Hicksian willingness to pay for the specified change in water quality. More formally, this measure of benefits would be defined as

$$\Delta\tilde{H}CS = E[TC^c, \bar{V}, \bar{\alpha}(C_N)] - E[TC^a, \bar{V}, \bar{\alpha}(C_N)]$$

$$- \{E[TC^c, \bar{V}, \bar{\alpha}(C_O)] - E[TC^a, \bar{V}, \bar{\alpha}(C_O)]\} \qquad (7.7)$$

where

$\Delta\tilde{H}CS$ = Hicksian compensating surplus for a quality change

TC^c = the choke price (travel cost) for a trip to the site*

TC^a = the actual price for a trip to the site

$\bar{\alpha}(C_j)$ = the demand parameter functions implied by the generalized travel cost model for characteristic C_j with J = N designating the new level and O the old level.

As noted, weak complementarity implies that $E[TC^c, \bar{V}, \bar{\alpha}(C_N)]$ will equal $E[TC^c, \bar{V}, \bar{\alpha}(C_O)]$ because there is assumed to be no demand for the characteristic when the demand for the site's services is zero. Mäler's [1974, pp. 183-186] argument that changes in environmental quality could not be valued without the restrictions implied by weak complementarity did not consider the possibility of observing behavior over a sufficient range of variation in environmental quality to estimate the $\bar{\alpha}(\cdot)$ functions.†

In summary, three measures of the benefits an individual receives from using the services of a recreation site will be estimated--the Marshallian consumer surplus with and without the assumption of a finite choke price and the Hicksian compensating surplus based on Hausman's quasi-expenditure function. These estimates will be developed for each of the sites under their existing

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*In the special case of the semi-log demand, an arbitrary upper bound on price would be required. We used the maximum travel cost experienced at each site.

†Mäler [1974, pp. 183-186] also appears to be suggesting that weak complementarity is an essential requirement for the use of a private good demand function in the valuation of a commodity that does not exchange on a market. This would seem to preclude the use of our user value as well. However, it appears to be the result of an implicit assumption that we cannot observe variation in the private good's demand with water quality. Further conceptual analysis will be required to relate Mäler's analysis with the approach implied by Hausman's analysis.

conditions to gauge the implications of selecting the different versions of the generalized travel cost model, (i.e., versions resulting from changes in sample composition and model specification).

7.3 SELECTING A MODEL FOR VALUING WATER QUALITY IMPROVEMENTS

This section illustrates the effects that the selection of an estimator and a model can have on the valuation of a site's services. It provides the basis for evaluating whether or not the approximations required in developing data for the model and the judgments required in selecting a final specification are important to the model's end uses.

The diversity of estimates and absence of potential explanations for them, from either a priori theory or the features of the data, severely complicates the analysis. Thus, some of the decisions are based largely on judgment. Where these judgments were necessary, they are documented along with the rationale used. One important judgment was using the effects of site attributes including water quality and measures of the activities on the intercept (α_0) and travel cost (α_1) parameters as the basis for selecting a final model. The effect of income on demand was excluded because it could not be measured precisely in a large number of the site demand functions.

Table 7-1 presents the final model that will be used to evaluate the effects of water quality changes and the activity mix. This model was estimated with the smallest sample of sites--the original sample of 22 sites used in Desvousges, Smith, and McGivney [1983] for which the predicted intercepts were positive and within the range of actual estimates for the demand functions. For the other coefficients, the signs were also consistent with a priori expectations for key variables like water quality. Note also that there are different models

Table 7-1. Generalized Travel Cost Model with Activities and ML Estimates

Independent variables	Intercept (α_0)	Travel cost parameter (α_1)	Income parameter (α_2)
Intercept	-4.2×10^{-3} (-0.024)	-1.45×10^{-1} (-3.341)	-7.51×10^{-5} (-1.762)
SHORMILE	5.94×10^{-4} (0.782)	3.14×10^{-5} (1.079)	-1.44×10^{-7} (-3.658)
MULTI+ACC	-3.88×10^{-2} (-1.071)	2.90×10^{-3} (1.534)	1.47×10^{-7} (0.112)
ARSIZE	1.46×10^{0} (1.030)	3.34×10^{-2} (0.388)	1.03×10^{-4} (2.890)
DOM	1.94×10^{-2} (2.076)	1.10×10^{-3} (2.623)	-7.35×10^{-8} (-0.177)
DOV	-6.47×10^{-5} (-2.077)		
DOM*Boat		-1.05×10^{-3} (-2.306)	-5.62×10^{-7} (-1.141)
DOM*Fish		-1.50×10^{-3} (2.157)	6.53×10^{-7} (2.461)
COLD			7.47×10^{-5} (6.541)
CGP		2.84×10^{-2} (1.041)	3.56×10^{-5} (3.189)
STOCK			

SHORMILE = Total shoremiles at the site during peak visitation period.

MULTI+ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

DOM*BOAT = Interaction between the activity measure for boating and the mean level of dissolved oxygen.

DOM*FISH = Interaction between the activity measure for fishing and the mean level of dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

STOCK = Qualitative variable indicating presence of an on-going fish-stocking program (=1).

specified for each parameter. This enables the “best” model to be matched with each coefficient.

Before discussing the water quality and activity values from the final model estimates using the 22-site sample, the poor quality of the model's performance with the larger sample of sites merits a brief explanation. Models estimated with the 42-site sample generated predictions that contradicted observed behavior. For example, it predicted negative values for the intercept under the existing water quality conditions coefficient which would imply very low levels of use at the sites with a zero travel cost and “average” income levels. The predicted values were also well outside the range of most of the original estimated intercepts for the site demand functions. A few large outlying estimates for the intercepts induced this response. The models for the intercept with the 33-site sample also predicted large negative intercepts in a majority of their second-stage equations. Thus, a similar pattern of predictions would emerge in this case as well. This performance is particularly disappointing because the additional data were collected, in part, to expand the number of sites for use in estimating the model.

The values of water quality improvements are calculated for a representative user for each of the 22 sites used in estimating the final model. Table 7-2 describes the features assumed for these representative users at each site along with measures of their physical characteristics and indexes of the mix of activities. The representative users are based on the mean travel cost and incomes of the surveyed respondents from each site. The choke price or upper limit to the integral defined in Equation (7.1) is the highest travel cost from the sample of site users (maximum travel cost in Table 7-2). Water quality conditions were assumed to be uniform for all sites at the dissolved oxygen

Table 7-2. Characteristics of Representative Individual and Site for Benefit Scenarios

Site name	Site No.	Representative individual characteristics: Income	Travel cost	Maximum travel cost	Site characteristics: SHOR-MILE	ARSIZE	MULTI + ACC	COLD	CGP	Boat	Fish
Arkabutla Lake, MS	301	13,184	20.04	209.35	134	0.6356	22	0	0	0.135	0.554
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	10,409	3.04	70.01	96	0.3270	8	0	1	0.196	0.826
Belton Lake, TX	304	17,279	33.18	302.86	136	0.7672	17	0	1	0.469	0.453
Benbrook Lake, TX	305	19,135	30.23	344.44	37	0.6755	7	0	1	0.166	0.417
Blakely Mt. Dam, Lake Ouachita, AR	307	17,144	45.39	286.03	690	0.5864	16	1	1	0.495	0.453
Canton Lake, OK	308	17,392	32.30	106.16	45	0.8966	9	1	1	0.466	0.636
Cordell Hull Dam and Reservoir, TN	310	15,491	29.65	184.35	381	0.4241	16	1	0	0.284	0.500
DeGray Lake, AR	311	19,235	42.04	210.48	207	0.5346	18	0	0	0.480	0.560
Grapevine Lake, TX	314	19,309	38.45	307.28	60	0.7146	9	0	1	0.372	0.309
Greers Ferry Lake, AR	315	15,890	54.16	451.00	276	0.8892	39	1	1	0.610	0.404
Grenada Lake, MS	316	9,199	24.57	207.05	148	0.7445	23	0	1	0.077	0.590
Hords Creek Lake, TX	317	16,263	39.46	304.01	11	0.4163	3	0	0	0.305	0.424
Melvern Lake, KS	322	18,087	31.48	130.50	101	0.5684	6	0	1	0.375	0.696
Millwood Lake, AR	323	18,630	37.62	309.24	65	0.7980	30	0	1	0.145	0.818
Mississippi River Pool No. 6, MN	325	19,589	52.23	843.86	55	0.7855	1	1	0	0.539	0.579
New Savannah Bluff Lock & Dam, GA	329	12,609	18.65	157.36	32	0.0123	0	0	0	0.021	0.553
Ozark Lake, AR	331	12,654	58.71	457.44	173	0.2700	13	0	0	0.093	0.333
Philpott Lake, VA	333	14,268	26.09	268.76	100	0.4229	14	1	1	0.429	0.286
Proctor Lake, TN	337	17,510	46.08	172.41	27	0.8780	4	0	0	0.538	0.654
Sam Rayburn Dam & Reservoir, TX	339	19,515	40.23	155.30	560	0.9325	27	0	1	0.471	0.729
Sardis Lake, MS	340	13,141	36.08	429.20	110	0.5934	16	0	1	0.310	0.504
Whitney Lake, TX	344	18,688	35.40	303.62	170	0.9359	17	0	1	0.436	0.538

SHORMILE = Total shoremiles at the site during peak visitation period.

ARSIZE = Total water area plus total land area.

MULTI + ACC = The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

CGP = Qualitative variable indicating extreme congestion during maximum usage periods.

level assumed in Desvousges, Smith, and McGivney [1983] to correspond to boatable water conditions (i.e., DOM = 45 in percent saturation).

Table 7-3 reports the Marshallian and Hicksian (based on Hausman's quasi-expenditure function) measures of the representative user's values of these sites with boatable water quality. The benefit estimates clearly indicate the sensitivity of these calculations to both the estimator used and the model specifications. For example, the estimates of the Marshallian consumer surplus for the original OLS differ substantially from the ML estimates. The benefit estimates derived from the site demands using the ML estimator were reduced by at least one-half of those based on the OLS estimates.* When the revised specification that includes activity mix for the second-stage models is included in the comparison, the differences are even more pronounced. Most estimates declined, but some are a small fraction of the estimates based on the original model. The wide range of values from the model including the activity mix and the low levels of some of these estimates in relation to observed data on travel and time costs for these sites suggest that this framework is an implausible description of recreationist behavior.

As would be expected on a priori grounds, the Marshallian and Hicksian measures of the willingness to pay are quite close for all estimators and models. However, on a priori grounds, the compensating variation would be ex-

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*The only exception was site No. 315, Greers Ferry Lake, which has several characteristics that fall near, or at, the upper bound of the range of values for site characteristics. This resulted in the smallest predicted coefficient for the travel cost variable, -.00127, which is approximately one-hundredth the size of the predictions for this parameter for the other sites and nearly as much smaller than the original ML estimate for this parameter (i.e, -.0287) for the site that was based on the behavior of the surveyed respondents. Consequently, this estimate will be regarded as an outlier and deleted from our comparisons of the estimated benefits with water quality improvements.

Table 7-3. A Comparison of Representative User's Value of Water-Based Sites with Boatable Water Quality--1977 Dollars

Site name	Site No.	Original model--OLS M	Original model--OLS H	Original model--ML M	Original model--ML H	Final model activities--ML M	Final model activities--ML H
Arkabutla Lake, MS	301	233.85	234.11	141.88	142.44	16.24	20.41
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	200.15	195.57	93.26	93.38	22.04	22.87
Belton Lake, TX	304	201.48	201.61	49.07	49.15	36.57	36.58
Benbrook Lake, TX	305	214.55	210.33	30.43	30.46	24.37	24.37
Blakely Mt. Dam, Lake Ouachita, AR	307	98.86	98.89	31.13	31.13	27.42	29.93
Canton Lake, OK	308	126.88	123.46	21.78	21.80	65.97	66.24
Cordell Hull Dam and Reservoir, TN	310	165.87	165.96	95.81	95.91	4.81	5.29
DeGray Lake, AR	311	162.24	162.34	81.76	81.90	0.53	1.31
Grapevine Lake, TX	314	189.06	182.16	20.18	20.20	17.61	17.62
Greers Ferry Lake, AR[a]	315	163.77	163.99	1,958.85	2,108.56	-	-
Grenada Lake, MS	316	205.87	206.07	82.18	82.40	194.29	194.88
Hords Creek Lake, TX	317	187.52	188.67	15.25	15.26	0.20	0.35
Melvern Lake, KS	322	152.42	150.14	26.23	26.24	3.32	3.32
Millwood Lake, AR	323	239.06	239.42	294.26	297.54	317.46	320.24
Mississippi River Pool No. 6, MN	325	144.16	149.84	2.29	2.29	0.27	0.27
New Savannah Bluff Lock & Dam, GA	329	232.87	235.83	55.31	55.32	0.66	1.54
Ozark Lake, AR	331	130.02	130.07	76.84	76.80	0.23	0.55
Philpott Lake, VA	333	227.53	227.68	86.13	86.27	36.13	36.17
Proctor Lake, TN	337	136.84	138.69	4.15	4.15	0.25	0.25
Sam Rayburn Dam & Reservoir, TX	339	117.69	117.77	75.47	75.63	141.71	190.85
Sardis Lake, MS	340	198.72	198.85	46.52	46.58	15.32	15.32
Whitney Lake, TX	344	187.66	187.77	35.58	35.63	70.38	70.47

M = Marshallian consumer surplus.

H = Hicksian compensating variation.

[a]This site has several site attributes at the upper bounds of the range which causes unreliable results. This site is deleted from any further calculations of benefits.

pected to be less than the Marshallian consumer surplus. In contradiction to theory, the estimates of the compensating variation always exceed the Marshallian estimates. This contradiction is attributable to the inexact correspondence to the theoretically proper definitions of each measure because an arbitrary upper bound was used for the choke price to approximate the theoretical counterparts.

In summary, these results indicate that estimators and the specification of the second-stage model do affect the valuation estimates. Moreover, the differences are clearly large enough to affect policy decisions. That is, according to the original criteria--whether the judgments used in composing a model would lead to differences in the resulting estimates that would be sufficient to affect either a policy decision or an evaluation of the importance of a theoretical argument--the answer is clearly "yes." Chapter 8 explores the implications of these findings for policy decisions in more detail. The original specification of the generalized travel cost model used in Desvousges, Smith, and McGivney [1983] continues to provide more reliable predictions of behavior than the expanded version of the model which includes the mix of activities. The original model is sensitive to the estimator used, but the range of the predictions does not seem to suggest the model is implausible. Unfortunately, this is not the case for the expanded model which produces many implausible predictions based on the existing conditions scenarios for the 22 recreation sites.

7.4 VALUING WATER QUALITY IMPROVEMENTS

This section discusses two important issues: the sensitivity of the estimated benefits of improving water quality and the implications for benefits estimation for the model that includes the role of activities at a recreation site.

To gauge the sensitivity of benefits estimates, a three-way classification based on differences in model, statistical estimator, and benefits measures is used for one sample of Corps of Engineers recreation sites. The evaluation of activities in the model focuses on general issues rather than presenting specific empirical estimates (see Appendix B for a sample of estimates).

7.4.1 Sensitivity of the Estimated Benefits of Improved Water Quality

The estimated benefits from improving water quality are described for two increments: from boatable to fishable (i.e., a change in dissolved oxygen from 45 to 64 percent) and boatable to swimmable (a change from 45 to 83 percent). The value of increments is based on illustrative scenarios instead of the actual water quality levels. In each scenario, each site is assumed to have a baseline water quality at boatable conditions, and the generalized travel cost model is used to predict the values of going from the baseline to fishable water quality, and then to swimmable water quality. The scenarios are usec instead of the actual water quality values because there was inadequate variation across the sites with the majority having dissolved oxygen levels in the swimmable range (above 80 percent saturation). Consequently, these estimates illustrate the model's capabilities, but they do not represent actual changes at any site.

Table 7-4 presents the estimated benefits of improved water quality based on 21 Corps of Engineers sites. The sample size was reduced by one with the ommission of the outlier observation (Site 315). For each water quality scenario presented, there are three sources of variation in the estimates of the value of a water quality improvement--estimator, model, and benefit measure (Marshallian or Hicksian). It is easier to evaluate these three for any

Table 7-4. A Comparison of the Benefit Estimates for Water Quality Improvements in 1977 Dollars

		Boatable to fishable						Boatable to swimmable					
		Original model--OLS		Original model--ML		Model with activities		Original model--OLS		Original model--ML		Model with activities	
Site name	Site No.	M	H	M	H	M	H	M	H	M	H	M	H
Arkabutla Lake, MS	301	104.57	288.57	31.20	58.74	11.09	12.16	274.20	825.78	70.19	145.23	29.88	31.71
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	39.97	634.14	29.63	59.66	9.34	7.49	89.45	101.04	68.84	166.66	22.84	17.59
Belton Lake, TX	304	115.84	300.12	10.27	20.02	11.56	14.17	331.45	1,621.23	22.78	48.86	26.85	34.02
Benbrook Lake, TX	305	124.64	136.99	6.63	13.27	23.77	19.09	366.68	411.22	14.75	32.75	71.83	54.26
Blakely Mt. Dam, Lake Ouachita, AR	307	49.54	128.82	5.19	23.08	6.31	7.56	131.73	602.18	11.36	79.37	14.19	17.18
Canton Lake, OK	308	42.83	1,433.42	4.99	8.97	7.01	4.26	101.59	117.08	11.15	21.80	15.26	9.24
Cordell Hull Dam and Reservoir, TN	310	68.75	192.50	17.47	58.99	2.65	2.69	173.75	856.19	38.67	175.42	6.77	6.76
DeGray Lake, AR	311	82.72	202.36	11.77	33.12	0.07	0.16	218.39	865.68	25.63	82.76	0.15	0.36
Grapevine Lake, TX	314	114.12	1,237.02	3.98	8.25	14.75	18.12	329.63	363.41	8.77	20.13	42.41	55.88
Grenada Lake, MS	316	99.16	203.26	20.54	33.31	170.34	103.85	262.04	726.10	46.61	81.52	500.15	277.14
Hords Creek Lake, TX	317	112.35	124.41	3.13	7.04	0.13	0.22	321.87	363.12	6.93	17.88	0.36	0.59
Melvern Lake, KS	322	56.21	64.00	35.71	12.05	0.57	[a]	136.35	158.66	12.70	30.28	1.25	[a]
Millwood Lake, AR	323	155 73	269.99	34.65	87.84	[a]	[a]	461.81	945.56	76.53	207.76	7.81	[a]
Mississippi River Pool No. 6, MN	325	100.17	115.69	0.40	0.86	[a]	[a]	300.51	355.39	0.87	2.07	[a]	[a]
New Savannah Bluff Lock & Dam, GA	329	84.92	95.35	13.25	569.00[b]	0.50	0.98	209.64	238.68	29.94	38.35	1.36	2.59
Ozark Lake, AR	331	94.66	252.48	6.91	37.69	0.47	1.05	291.05	2,002.09	15.48	111.68	1.91	4.26
Philpott Lake, VA	333	117.99	355.79	17.62	39.91	25.40	30.21	328.58	2,422.17	39.38	102.62	69.59	86.85
Proctor Lake, TN	337	68.93	78.33	0.63	1.58	[a]	[a]	178.22	207.01	1.83	3.79	[a]	[a]
Sam Rayburn Dam & Reservoir, TX	339	49.30	100.18	12.87	29.90	[a]	[a]	122.62	317.87	28.12	73.07	[a]	[a]
Sardis Lake, MS	340	128.98	321.33	9.76	19.04	7.73	4.61	398.58	1,923.47	21.73	46.94	19.36	10.96
Whitney Lake, TX	344	109.78	295.36	7.36	14.15	14.58	12.16	315.02	1,730.25	16.29	34.24	32.79	27.02

M = Marshallian consumer surplus.

H = Hicksian surplus calculated using Hausman's quasi-expenditure function.

[a]Negative predictions for value of water quality improvement.

[b]This large discrepancy in a counterintuitive direction arises from a change in the sign of the predicted coefficient of income for the scenario boatable to swimmable from positive to negative. This parameter plays a key role in the Hausman quasi-expenditure function (see Equation (6.6) in the text).

one site because characteristics of the area and the features of the representative user can be held constant.*

The results in Table 7-4 confirm what would have been expected based on the earlier evaluation of the models in valuing a site's services at boatable water quality. The estimated benefits for water quality have a substantial range--with the original OLS-based model predicting estimates that are well above the ML using the same model specification. For example, the estimated Marshallian consumer surplus for improving water quality to fishable levels shows large differences depending on the estimator used (OLS vs. ML). For many of the sites, the OLS-based estimates are one order of magnitude larger than those based on the ML estimator. In at least six cases, the differences are even more dramatic--e.g., for Mississippi River Pool No. 6, the OLS estimates are $100 per year, and the ML estimates are $0.40. The same general picture emerges for the estimated benefits for attaining swimmable water quality. While neither set of estimates should be regarded as the "true" ones, the dramatic differences suggest caution when using the model to support policy.

Estimates of the value of water quality are also quite dependent on the benefit concept used. In contrast to the proximity between the Marshallian and Hicksian measures for the case of valuing a site's services (i.e., a price change from the maximum observed travel cost to the average travel cost), these measures are quite different in their estimation of the willingness to pay

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*The Marshallian surplus based on the indefinite integral is not reported here. It is based on one coefficient--the parameter for the travel cost--and will not vary across individuals with variations in either the travel cost or income. Moreover, the ML estimates indicated that water quality primarily affected the intercept parameter and not the parameter for travel cost. The implications of the selection of a benefit measure will be considered further in Chapter 8.

for water quality improvements. Once again the Hicksian measure is usually greater than the Marshallian estimate, though there are some exceptions in the case of the less reliable models with the activity indexes included in the second-stage equations.

7.4.2 Incorporating the Role of Activities

The results in Table 7-4 provide further evidence of the problems with the second-stage model that includes the activity indexes. It appears more unstable than the other models, predicts coefficient values outside the feasible economic region, and leads to negative estimates of the value of water quality. Yet, this assessment relies largely on judgment and is not based on any statistical tests or strong theoretical evidence. Thus, some attempt was made to consider the model's implications for differences in activity mix on the valuation of water quality improvements. A wide array of scenarios--varying the mix of activities by specifying values for the activity indexes--were constructed to address the activity mix question. In all cases, these scenarios were completely ineffective.

There seemed to be several reasons for this failure. The most important of these may well be the data inadequacies addressed earlier. Another problem stems from the somewhat contradictory formulation of the scenarios. Based on our theoretical analysis, the activity mix should not change when there are changes in site attributes. This would imply that boating should be the only feasible activity for the low water-quality levels. When benefits are calculated under the assumption of the existing activity mix at each site, it could include both boating and fishing. However, the assumed baseline level of water quality was postulated to be boatable water quality. Of course, for most sites, this is not the actual condition so recorded behavior and assumed water quality are inconsistent. When using scenarios in the model, either the character of the fishing must be assumed to change (and there is no basis for reflecting this change in the model) or scenarios must be designed substantially outside the range of experience with these sites. Neither approach is desirable.

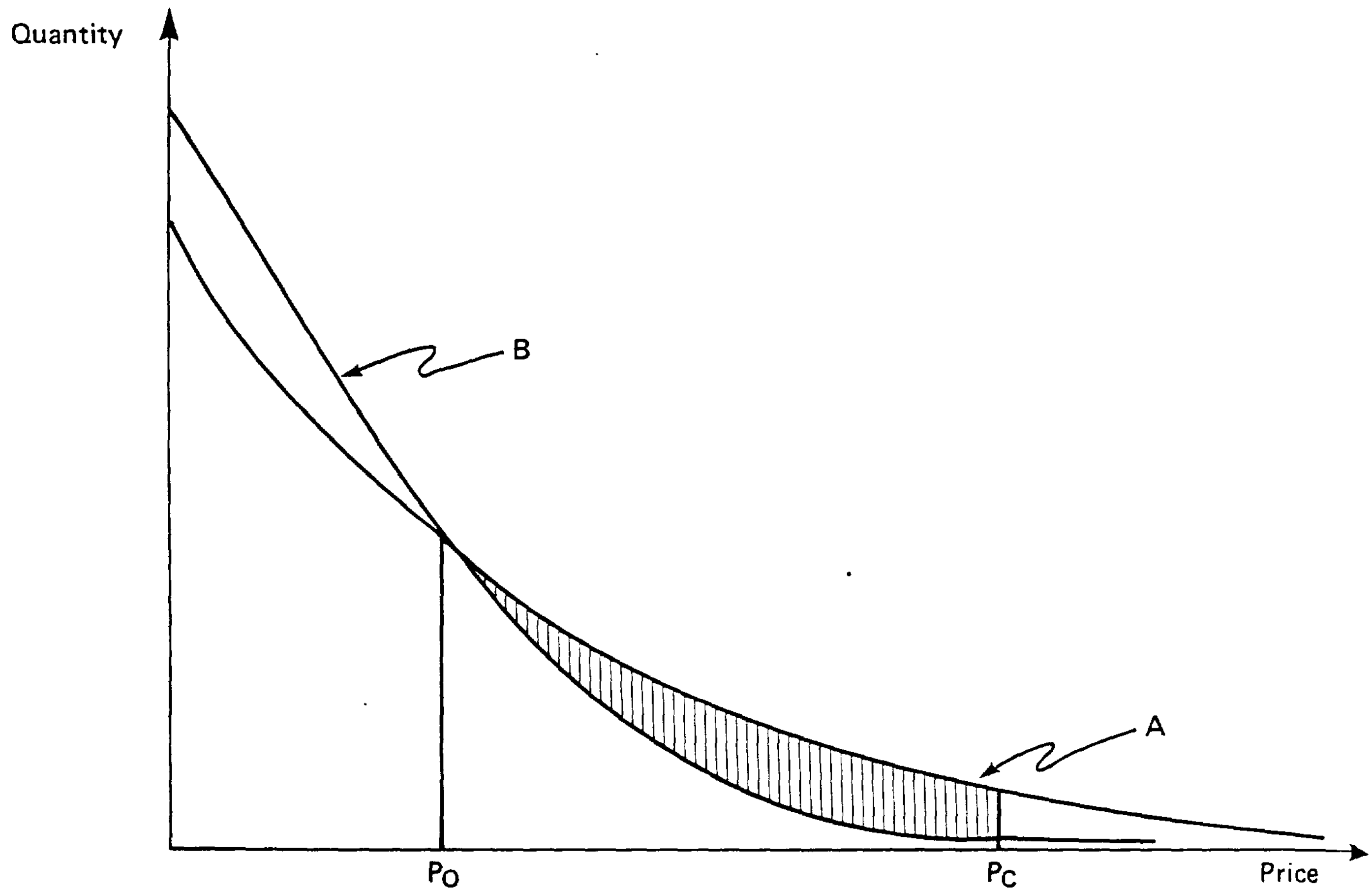
For example, in the construction of a benefit scenario, initial and final values for water quality and the activity variables must be specified. Two specifications of the activity indexes were possible: (1) constant at specified levels before and after the water quality changes but consistent with what the before and after water quality would permit, and (2) different specified values before and after the water quality change. Use of the first method with our baseline of boatable water quality conditions assumes that there was no fishing at a site, but that boating activities occurred, and then estimates the value of a water quality improvement from boatable to fishable. The second scenario has the same specification for the initial point (i.e., at boatable water quality only boating activities take place), but when water quality improves to fishable, fishing is assumed to occur at some specified level.

Scenarios based on adjustments to the actual conditions or other modifications in the activity indexes generally lead to predicted coefficients outside the economic range and benefit estimates that were either negative or inconsistent with our a priori expectations for the effects of activities. For example, those sites specified to support increased fishing after a water quality improvement leading to fishable water were found to have either negative benefits or benefits less than those that did not support any fishing.

Examination of the coefficients in the models for the travel cost parameter from Table 7-1) reveals that increases in activity indexes imply reductions in the measured benefits of a water quality improvement. As either index approaches unity, the contribution of activities tends to offset the effect of direct water quality on the travel cost parameter (α_1). Benefits are then primarily determined by the shift in the intercept of the demand function, which is the result of the joint effects of water quality through predictions of α_0 and water quality and activities on the income parameter (α_2). These can lead to reduced benefits, because of the configuration of negative signs in this second-stage equation. Figure 7-4 illustrates the difficulty, with demand curve A designating the before, and B the after water quality scenarios improvement. For illustration, selected scenarios are presented in Appendix B.

The decisions required to compose benefit scenarios for the generalized travel cost model illustrate another important problem with the attempt to include activities in the model. The implicit assumption (discussed in Chapter 2) that all activities can be undertaken at a site regardless of the level of site characteristics is quite limiting. Improvements in characteristics served to increase the productivity of a site's services in the production of recreational service flows (e.g., fishing or swimming). While it is possible that these improvements were of differential magnitudes across activities, the model did not allow for the characteristics affecting the activity choices. The choices to fish or swim were given exogenously.

Clearly, the process of developing specifications for our scenarios illustrates how limiting this assumption can be. On theoretical grounds, the model is inconsistent with the second type of scenario. The specific mix of recreational activities whose derived demands are aggregated must be exogenous to the levels of the site characteristics. If it is not, then the formal model is inadequate, and the selection of activities and sites must be considered part of a simultaneous decision process.



P_O = Existing Price
P_C = Maximum Price

Figure 7-4. Illustration of reduction in consumer surplus with water quality improvement.

To provide a preliminary investigation of the potential for simultaneity in the selection of activities and sites, several simple regression models were estimated with the activity indexes specified to be a function of site attributes including water quality. Results are reported in Appendix C. The results were not supportive of the simultaneity argument, but the available indexes offer a weak basis for evaluating the hypothesis. A more substantial test would require information on individual choices rather than indexes of average selections for all surveyed users.

7.5 SUMMARY

This chapter has reported a series of benefit estimates for the expanded models developed in Chapter 6 as well as the original generalized travel cost model described in Desvousges, Smith, and McGivney [1983]. While the theoretical analysis in Chapter 2 indicated that it was conceptually feasible to model the effects of the activity mix on the value of improved water quality, the empirical evaluation of the water quality benefits was unsuccessful for at least two reasons. First and probably most important, the available data on each recreationist's activities while at a site are simply not up to the requirements of the theoretical model. The crude indexes of average participation in two activities--boating and fishing--during trips to the site did not permit the empirical analysis to distinguish participation patterns of the recreationists using the same site, but only accounted for variations across sites. Moreover, the recording of the participation decisions did not imply that they were mutually exclusive selections. A respondent could report participating in several activities while visiting a site, but the available information does not indicate the amount of time devoted to each. Given these limitations, it is not surprising to find that our estimated second-stage models were sensitive to the specification used.

The second potential reason for the failure of the model to distinguish the benefits of water quality improvements stems from theoretical limitations of the model: the inability to isolate the role of activities undertaken onsite in the individual's valuation of site services. The model assumes that changes in site characteristics do not affect the decision to participate in particular activities with only the amount of use of the site being affected. Once this assumption is relaxed, site selection and activity choice become endogenous variables that must be modeled in a single system. Our model avoided this problem by specifying the set of activities undertaken by the individual in advance, avoiding the prospect for changes of the mix. This is an important limitation in the sense that the full range of individual choices cannot be considered.

Despite these limitations, the empirical analysis has provided several extensions to the earlier work. Our revised analysis has considered the effects of estimator, model, and valuation method for water quality benefits. The refinements in estimator and benefit measure have improved our understanding of the potential sources of discrepancies in benefit estimates and indirectly clarified the appropriate definition of the Hicksian valuation of a quality change.

Finally, the analysis of activities has pointed out the types of data that would be critical in improving the empirical estimation. Ideally, a diary detailing the time spent at each activity and whether some activities were undertaken jointly (i.e., in the same time intervals) would be necessary. These insights on data requirements may prove helpful to those designing large-scale recreation surveys such as the 1977 Federal Estate Survey.

CHAPTER 8

BENEFITS VARIABILITY: AN EVALUATION

8.1 INTRODUCTION

This chapter examines the results of this and other recent benefits research to evaluate the effects of variability among benefit estimates on the use of benefit-cost analysis. In developing this evaluation, the following sections address several important questions:

- What are the sources and extent of variability in benefits?
- What are the practical implications of the variability in benefits for benefit-cost analysis?
- How does the variability in benefits estimates compare with that in other studies?
- What recommendations are possible for analysts who conduct benefit-cost analyses?

In particular, Section 8.2 examines the amount of variability attributable to differences in the measures of well being, the statistical estimator, and the units of use employed in this study; Section 8.3 compares benefit estimates across 21 recreation sites and evaluates them in light of the differences among site and user characteristics; and Section 8.4 considers the relationship among benefits variability, benefit-cost decision criteria, and the treatment of uncertainty in the benefit-cost assessment process. In addition, Section 8.5 offers several hypothetical benefit-cost assessments to illustrate the effects of benefits variability on conducting such analyses; Section 8.6 compares the benefits estimated during this study with those estimated in two recent studies; and, finally, Section 8.7 concludes the chapter by briefly addressing the issue of benefits transfer--the process of transferring benefits estimated for one site to a policy analysis relating to another site.

8.2 SOURCES AND EXTENT OF VARIABILITY

This section examines the sources and extent of variability in the estimated benefits of improved water quality for particular recreation sites. In particular, it evaluates how much of the variation can be attributed to the differences in three factors:

- Unit of use--per trip or per day.
- Statistical estimator--ML or OLS
- Welfare measure--Hicksian or Marshallian

The first source, the unit of use, arises when the benefits are rescaled from the unit used in the original analysis (a season) to the two alternative units that are commonly used in recreation benefits studies (visitor trip or visitor day). Frequently, these units are also desired by analysts and policymakers for planning or policy evaluation (see Dwyer, Kelley, and Bowes [1977], and Loomis and Sorg [1982]). The second source of variability, the statistical estimator, refers to the two procedures that were used to estimate the first stage of the travel cost demand equation. The third source, the welfare measure, stems from the alternatives basis used to measure the change in well being. The following subsections address each of the factors as sources of the variability in the estimations of benefits.

8.2.1 Variability Due to Unit of Use

As a first step in evaluating the variability due, to unit of use, Table 8-1 displays the estimated values of improved water quality based on using the ML statistical estimation procedure. Table 8-2 provides similar estimates for the OLS statistical estimation procedure. Within each table, estimates are provided for two levels of water quality, two units of use, and the alternative welfare measures.

Table 8-1. Value of Water Quality per Unit of Use: ML Estimates

Site name	Site No.	Value per trip: Boatable to fishable M1	M2	H	Value per trip: Boatable to swimmable M1	M2	H	Value per visitor day: Boatable to fishable M1	M2	H	Value per visitor day: Boatable to swimmable M1	M2	H
Arkabutla Lake, MS	301	3.06	5.78	10.88	5.69	13.28	26.89	8.05	15.21	28.63	14.97	34.95	70.76
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	1.57	4.36	8.77	2.97	10.12	24.51	1.26	3.49	7.02	2.38	8.10	19.61
Belton Lake, TX	304	0.76	1.71	3.34	1.46	3.80	8.14	0.38	0.86	1.69	0.74	1.92	4.11
Benbrook Lake, TX	305	0.40	2.88	5.77	0.79	6.41	14.24	0.16	1.18	2.36	0.32	2.62	5.81
Blakely Mt. Dam, Lake Ouachita, AR	307	1.37	1.21	5.37	2.61	2.64	18.46	0.27	0.23	1.04	0.51	0.51	3.58
Canton Lake, OK	308	0.28	1.09	1.95	0.54	2.42	4.74	0.04	0.34	0.61	0.17	0.75	1.48
Cordell Hull Dam and Reservoir, TN	310	3.79	3.06	10.35	7.01	6.78	30.78	2.71	2.19	7.39	5.01	4.84	21.99
DeGray Lake, AR	311	2.66	2.45	6.90	4.98	5.34	17.24	0.72	0.67	1.88	1.36	1.46	4.70
Grapevine Lake, TX	314	0.43	0.63	1.76	0.83	1.39	3.20	0.12	0.18	0.49	0.23	0.39	0.89
Grenada Lake, MS	316	1.85	3.21	5.21	3.51	7.28	12.74	1.95	3.38	5.48	3.69	7.66	13.41
Hords Creek Lake, TX	317	0.60	0.71	1.60	1.17	1.58	4.06	0.24	0.28	0.63	0.46	0.62	1.60
Melvern Lake, KS	322	0.50	1.33	2.80	0.97	2.95	7.04	0.19	0.49	1.04	0.36	1.09	2.61
Millwood Lake, AR	323	5.90	6.19	15.69	10.73	13.67	37.10	3.17	3.33	8.44	5.77	7.35	19.95
Mississippi River Pool No. 6, MN	325	0.21	0.08	0.18	0.42	0.18	0.43	0.10	0.04	0.08	0.19	0.08	0.20
New Savannah Bluff Lock & Dam, GA	329	2.41	2.28	98.10	4.53	5.16	6.61	20.08	19.00	817.50	37.75	43.00	55.08
Ozark Lake, AR	331	7.05	1.41	7.69	12.71	3.16	22.79	5.26	1.05	5.74	9.49	2.36	17.00
Philpott Lake, VA	333	2.54	3.04	6.88	4.77	6.79	17.69	1.60	1.91	4.33	3.00	4.27	11.13
Proctor Lake, TN	337	0.21	0.15	0.29	0.42	0.34	0.70	0.57	0.04	0.08	0.11	0.09	0.14
Sam Rayburn Dam & Reservoir, TX	339	1.28	3.14	7.29	2.44	6.86	17.82	0.26	0.63	1.46	0 49	1.38	3.58
Sardis Lake, MS	340	1.37	1.50	2.93	2.61	3.34	7.22	0.49	0.54	1.05	0.94	1.20	2.59
Whitney Lake, TX	344	0.45	1.47	2.83	0.87	3.26	6.85	0.14	0.46	0.88	0.27	1.02	2.13

M1 = The Marshallian measure without an upper limit on the choke price.

M2 = The Marshallian measure with the maximum travel cost as a choke price.

H = The Hicksian compensating surplus based on Hausman's quasi-expenditure function.

Table 8-2. Value of Water Quality per Unit of Use: OLS Estimates

Site name	Site No.	Value per trip						Value per visitor day					
		Boatable to fishable			Boatable to swimmable			Boatable to fishable			Boatable to swimmable		
		M1	M2	H	M1	M2	H	M1	M2	H	M1	M2	H
Arkabutla Lake, MS	301	29.17	19.37	42.37	96.11	50.78	152.92	76.76	50.95	111.39	252.92	133.63	402.42
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	22.37	5.88	93.26	67.48	13.15	14.86	17.84	4.70	74.61	53.98	10.52	11.90
Belton Lake, TX	304	29.29	19.31	50.02	96.66	55.24	270.21	14.79	9.75	25.26	48.32	27.90	136.47
Benbrook Lake, TX	305	30.45	54.19	59.56	101.97	159.43	178.79	12.43	22.12	24.31	41.62	65.16	72.98
Blakely Mt. Dam, Lake Ouachita, AR	307	10.17	11.52	29.96	25.97	30.64	140.04	1.97	2.24	5.82	5.04	5.95	27.19
Canton Lake, OK	308	35.27	9.31	311.61	125.76	27.09	25.45	10.99	2.90	97.07	39.18	8.44	7.93
Cordell Hull Dam and Reservoir, TN	310	15.17	12.06	33.77	41.60	30.48	150.20	10.84	8.61	24.12	29.71	21.77	107.29
DeGray Lake, AR	311	22.41	17.23	42.16	67.64	45.50	180.35	6.11	4.69	11.49	18.43	12.40	49.14
Grapevine Lake, TX	314	30.36	18.12	196.35	101.56	52.32	57.68	8.43	5.03	54.54	28.21	14.53	16.02
Grenada Lake, MS	316	30.54	15.49	31.76	102.40	40.94	113.45	32.15	16.31	33.43	108.00	43.09	119.42
Hords Creek Lake, TX	317	26.35	25.54	28.28	106.94	73.15	82.53	10.37	10.05	11.13	42.10	28.80	32.49
Melvern Lake, KS	322	24.51	13.07	14.88	76.04	31.71	36.90	9.08	4.84	5.51	28.16	11.74	13.67
Millwood Lake, AR	323	43.66	27.81	46.96	174.14	82.47	168.85	23.48	14.95	25.25	93.62	44.34	90.78
Mississippi River Pool No. 6, MN	325	28.69	20.87	24.10	93.91	62.61	74.04	13.28	9.66	11.16	43.48	28.99	34.28
New Savannah Bluff Lock & Dam, GA	329	19.77	14.64	16.44	57.64	36.15	41.15	164.75	122.00	137.00	480.33	301.25	342.92
Ozark Lake, AR	331	19.82	19.32	51.53	57.84	59.40	408.59	14.79	14.42	38.46	43.16	44.33	304.92
Philpott Lake, VA	333	25.12	20.34	61.34	51.12	56.65	417.62	15.80	12.79	38.58	32.15	35.63	262.65
Proctor Lake, TN	337	33.84	12.77	14.51	118.47	33.00	38.34	9.25	3.49	3.96	32.37	9.02	10.47
Sam Rayburn Dam & Reservoir, TX	339	15.56	12.03	24.44	42.89	29.91	77.53	3.12	2.42	4.91	8.61	6.01	15.57
Sardis Lake, MS	340	27.73	19.84	49.44	89.67	61.32	836.29	9.94	7.11	17.72	32.14	21.98	299.75
Whitney Lake, TX	344	29.90	21.96	59.07	99.45	63.00	346.05	9.31	6.84	18.40	30.98	19.63	107.80

M1 = The Marshallian measure without an upper limit on the choke price.

M2 = The Marshallian measure with the maximum travel cost as a choke price

H = The Hickstan compensating surplus based on Hausman's Quasi-expenditure function.

To obtain the per-trip benefits, the seasonal estimates presented in Chapter 7 were rescaled using the average number of visits for each site.* The per-visitor day estimates were obtained by using the average length of stay at each of the sites, which can be found in Table 8-3. The average length of stay differs significantly across the 21 Corps of Engineers sites, with ranges from about 2 hours to more than 5 days. The presence of sites with average visit lengths of less than 1 day causes the benefits per visitor day for these sites (301, 316, and 329) to exceed their per-trip counterparts. Instead of using a standardized definition of an activity day, the per-day estimates are left in the average length reported for the sample of users at the sites. This decision does not change the. variability in benefits per visitor day for a site, but it can lead to a larger range of daily values when the estimates are compared across sites.

8.2.2 Variability Due to Welfare Measure

The extent of variability in the benefits estimates attributable to the choice of a welfare measure can be obtained by considering the range in estimates for a given estimator. For example, Table 8-1 shows two sites, New Savannah Bluff Lock & Dam and Benbrook Lake, with substantial differences in the estimated benefits, depending on the welfare measure used in calculating the benefits of attaining fishable water quality. For the New Savannah River

*An alternative basis for these calculations would have been the predicted number of trips at each water quality level. With the semi-log model, these estimates will be biased and tend to understate use. Actual use levels were generally associated with the higher values of the water quality variables, since most sites had these water qualities as their actual conditions. Consequently, the actual use was considered a better estimate of use under the improved water quality conditions for most sites.

Table 8-3. Average Length of Visit

Site name	Site No.	Visit length (days)
Arkabutla Lake, MS	301	0.38
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	1.25
Belton Lake, TX	304	1.98
Benbrook Lake, TX	305	2.45
Blakely Mt. Dam, Lake Ouachita, AR	307	5.15
Canton Lake, OK	308	3.21
Cordell Hull Dam and Reservoir, TN	310	1.40
DeGray Lake, AR	311	3.67
Grapevine Lake, TX	314	3.60
Grenada Lake, MS	316	0.95
Hords Creek Lake, TX	317	2.54
Melvern Lake, KS	322	2.70
Millwood Lake, AR	323	1.86
Mississippi River Pool No. 6, MN	325	2.16
New Savannah Bluff Lock & Dam, GA	329	0.12
Ozark Lake, AR	331	1.34
Philpott Lake, VA	333	1.59
Proctor Lake, TN	337	3.66
Sam Rayburn Dam & Reservoir, TX	339	4.98
Sardis Lake, MS	340	2.79
Whitney Lake, TX	344	3.21

site, the range is on the order of 40 to 1, while the range for Benbrook Lake is about 15 to 1. However, the ranges of benefits for attaining the same level of water quality are much smaller for the remainder of the sites. For 10 of the sites, the variability in benefits is 3 to 1 or less. The ranges for the remaining 8 sites fall between 4 and 6 to 1. Moreover, the Hicksian measures are always larger than the Marshallian measures. Unfortunately, the behavior of the two Marshallian estimates does not follow any consistent pattern.

The estimated benefits of attaining swimmable water quality based on the ML estimator also exhibit more pronounced variability than those for attaining fishable water quality. Some of this variability is attributable to the variability in benefits for achieving the lower water quality level that is also reflected in the estimates for the improvement from boatable to swimmable. That is, the estimated benefits of attaining swimmable water quality also include the benefits of attaining fishable water quality. Yet there is still an incremental increase in the variability of the benefits estimates for the higher water quality. For example, the highest estimates of swimmable water quality benefits for Benbrook Lake increase from 15 to 17 times more than the lowest estimate. Six of the remaining sites have ranges within 10 to 1, while the others have ranges of less than 5 to 1. Thus, the level of benefit variability increases for the estimates of attaining swimmable water quality, but, for the majority of the sites, the variability is still one order of magnitude or less.

A similar appraisal of the welfare measure’s contribution to benefits’ variability can be obtained by examining the estimates derived using the OLS estimator. Overall, the variability in the estimates made with this estimator appears to be less. For example, the ranges for 18 of the 21 sites are on the order of 3 to 1 or less for attaining fishable water quality. However, three of the sites--Lock and Dam No. 2, Canton Lake, and Grapevine Lake--show increases in levels of variability to greater than one order of magnitude. All three of these sites are different from the ones that showed the highest levels of variability using the ML procedure.

The variability in benefits of attaining swimmable water quality for the OLS-based estimates shows a similar pattern, although there are two important exceptions. The size of the estimated benefits for the Marshallian measure, M1, without an upper limit on the choke price--"the one over the coefficient" measure--increases relative to the other measure. This may in part reflect the total dependence of this measure on one of the estimated coefficients of the model. The second exception is the substantial increases--for example, on a per-trip basis increase of more than several hundred dollars for 5 of the sites. The worst case, Sardis Lake, has a range of benefits of almost two orders of magnitude. On a per-trip basis, the range for Sardis Lake extended from about $60 per trip for the Marshallian measure to over $800 per trip with the Hicksian measure.

8.2.3 Variability Due to Statistical Estimator

The extent of variability in the estimated benefits attributable to the choice statistical estimator also can be appraised by using the data in Tables 8-1 and 8-2. This appraisal holds the welfare measure constant while allowing the estimator to vary and requires one to compare across the tables rather than within each table. To simplify the exposition, only the benefits of attaining fishable water quality are discussed. Fortunately, the picture that could be drawn for the variability in the benefits of attaining swimmable water quality is very similar.

One central conclusion about the variability in benefit estimates across estimators emerges quite clearly: the variability is considerably greater than that across welfare measures. The conclusion holds true for all three measures of well being. For any given site, the OLS-based estimates are markedly larger than those based on the ML estimator. For example, several sites have Marshallian measures with ranges that are greater than two orders of magnitude. This occurs for Canton Lake, Mississippi River Pool No. 6, and Proctor Lake. The level of variability observed at Mississippi River Pool No. 6 is typical of the three sites. At this site, M1 ranges from $.21 per visitor trip based on the ML estimator to $28.69 for the OLS-based measure. The variability in the M2 welfare measure is no better with a range from $.08 per trip to almost $21 per trip. The case for the Hicksian-based measures is the same for this site.

Generally, the ranges associated with the estimators are wider than those associated with welfare measures for every site and all 3 welfare measures. A few sites have high estimates exceeding lows by about 5 to 1, but these are clearly the exception. High estimates 10 to 20 times greater than the lows are the rule, but several of the sites are in the 50 range. The level of variability is most pronounced for the M1 welfare measure, with the Hicksian measure next, and the M2 measure showing the least of the three. However, even with the measure that has the lowest variation (M2), there are three sites that have high estimates 50 times larger than low estimates.

In summary, the extent of variability in benefits estimates for a particular site is very sensitive to the choice of the statistical estimation procedure. This sensitivity is evident across all three measures of well being. Differences between high and low estimates for a given welfare measure at a given site were almost always on the order of one order of magnitude, and many of the sites had differences of at least one and one-half orders of magnitude.

8.3 COMPARISONS OF BENEFIT ESTIMATES ACROSS SITES

This section compares the estimated benefits of improved water quality across the 21 sites in this study and across three other recent benefits studies. The objective of this comparison is to appraise the sources of the variability in benefits estimates across the various Corps sites.

8.3.1 Comparison of Benefits Across the 21 Corps of Engineers Sites

Comparing benefits estimates of attaining fishable or swimmable water quality across sites is a considerably different task than comparing them for the same site. The estimated benefits from the generalized travel cost demand model can be expected to differ across sites if there are substantial differences among the characteristics of the sites and/or of their users. These kinds of differences would be reflected in different intercepts and slopes for the travel cost demand function which would yield different benefits estimates for each site. However, substantial differences across sites make it difficult to use a unit-day value measure of benefits--to represent the benefits of attaining fishable or swimmable water quality at any one site. Clearly, benefits estimated from a travel cost model designed specifically for the site, or from a contingent valuation survey for the site, either of which would account for site specfic characteristics, would be preferred.

Nevertheless, the extent of the variability across sites can be addressed from several perspectives: the variability for a given welfare measure, the variability for a given estimator, and the variability across estimator and welfare measure. As shown in Tables 8-1 and 8-2, which display the variability in the per visitor trip benefit estimates of attaining fishable water quality across the 21 sites, the Hicksian welfare measure has the widest range of estimated benefits of attaining fishable water quality. These estimates range from $.18 per visitor trip to $312 per visitor trip. The variability for the Marshallian measure (M2) is also considerable, with estimates ranging from $.08 per visitor trip to about $54. The Marshallian measure--M2 shows the least variability of the three welfare measures but its range is still broad, $.21 to $44.

The differences on a per-visitor day basis are comparable except for the importance of two sites. The ranges are especially sensitive to Arkabutla Lake and New Savannah Bluff Lock and Dam. With these two sites omitted, the ranges for all three welfare measures decrease substantially. For example, the range for the Hicksian measure declines to $.08 to $97 per visitor day. Without these two sites, the M1 benefits-estimates range is $.10 to $32 per visitor day, while that for M2 is $.04 to $22.

A similar picture for both units of use emerges for the variability for a given estimator and the variability across both estimator and welfare measures. Indeed, the range of all the benefit estimates across all dimensions in Tables 8-1 and 8-2 is only slightly greater than the variability for a given welfare measure or estimator. Broad ranges of benefits estimates across sites are present regardless of how they are appraised. Nevertheless, only two of the sites contribute a substantial share of this range.

To provide some perspective on this substantial variability of benefit estimates across sites, Table 8-4 shows the differences--expressed as a percent of the mean values--in the site characteristics and representative users across the 21 Corps of Engineers sites. As might be expected, the differences in both site and user characteristics are pronounced. For example, the travel cost measure ranges from Ozark Lake (Site 331) at 1.7 times the mean value

Table 8-4. Differences in Characteristics of Representative Individual and Sites for Benefit Scenarios

		Representative individual characteristics (percent of mean value)			Site characteristics (percent of mean value)						
Site name	Site No.	Income	Travel cost	Maximum travel cost	SHORMILE	ARSIZE	MULTI+ ACC	COLD	CGP	Boat	Fish
Arkabutla Lake, MS	301	.82	.58	.76	84	.97	1 47	0	0	.42	1 02
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	.65	.09	.26	60	.50	53	0	1.61	60	1 52
Belton Lake, TX	304	1.07	.97	1 10	86	1.17	1.13	0	1 61	1 45	83
Benbrook Lake, TX	305	1.19	.88	1.26	.23	1.03	47	0	1.61	51	77
Blakely Mt. Dam, Lake Ouachita, AR	307	1.06	1.32	1 04	4 34	.89	1.07	4.17	1 61	1.53	83
Canton Lake, OK	308	1.08	94	39	.28	1 36	60	4 17	1.61	1.44	1 17
Cordell Hull Dam and Reservoir, TN	310	96	.86	67	2 40	65	1 07	4.17	0	.88	.92
DeGray Lake, AH	311	1 19	1.22	.77	1 30	.81	1 20	0	0	1.48	1 03
Grapevine Lake, TX	314	1 20	1 12	1 12	38	1.09	60	0	1.61	1.15	57
Grenada Lake, MS	316	57	72	.75	93	1.13	1.53	0	1.61	.24	1 09
Hords Creek Lake, TX	317	1 01	1 15	1 11	07	63	20	0	0	94	78
Melvern Lake, KS	322	1 12	92	.48	64	.86	40	0	1.61	1 16	1 28
Millwood Lake, AR	323	1 15	1 10	1.13	41	1 21	2 01	0	1.61	45	1 51
Mississippi River Pool No. 6, MN	325	1.21	1 52	3.07	35	1.19	.07	4 17	0	1.66	1 07
New Savannah Bluff Lock & Dam, GA	329	78	54	.57	20	.02	00	0	0	.06	1 02
Ozark Lake, AR	331	.78	1 71	1 67	1 09	.41	87	0	0	29	.61
Philpott Lake, VA	333	88	76	98	63	64	93	4 17	1 61	1 32	53
Proctor Lake, TN	337	1 09	1.34	63	17	1.34	27	0	0	1 66	1 20
Sam Rayburn Dam & Reservoir, TX	339	1 21	1 17	57	3.52	1 42	1 80	0	1 61	1 45	1 34
Sardis Lake, MS	340	81	1 05	1 56	69	90	1 07	0	1.61	96	93
Whitney Lake, TX	344	1 16	1 03	1 11	1 07	1 42	0	0	1 61	1 35	99
Site Average		16,130	34 35	274 27	159	6575	15	24	62	324	543

SHORMILE Total shoremiles at the site during peak visitation period.

ARSIZE Total water area plus total land area.

MULTI + ACC The sum of the developed multipurpose onsite recreation areas and developed onsite access areas.

COLD Qualitative variable indicating presence of coldwater gamefish (=1).

CGP Qualitative variable indicating extreme congestion during maximum usage periods.

of $34.35 to Lock and Dam No. 2 on the Arkansas River Navigation System at 0.09 of the mean value. The differences in maximum travel costs are even larger, with Mississippi River Pool No. 6 having costs totaling more than three times the mean, while Lock and Dam No. 2 is only slightly more than one-fourth of the mean value.

The difference is even more pronounced for the various site characteristics. Hord's Creek Lake has a shoreline that is 7 percent of the mean value, while Lake Ouachita's shoreline is almost four and one-half times the mean value. Similar levels of variation are present in measures of size and access for the sites. Thus, some of the variability in benefits estimates across the 21 Corps of Engineers sites can be traced directly to the diversity among the site characteristics and users. Nevertheless, these differences explain none of the variability for a given site that was noted earlier.

8.4 BENEFITS VARIABILITY IN A BENEFIT-COST FRAMEWORK

This section explores the inextricable relationships between benefits variability and two central components of a benefit-cost analysis--the nature of the decision criteria and the treatment of uncertainty.

8.4.1 The Nature of the Decision Criteria

The basic decision rule, or criteria, in benefit-cost analysis is straightforward: undertake actions where anticipated benefits exceed the anticipated costs (Burkhead and Miner [1971]). This simple decision rule can take many forms, depending on the final objective of the analysis. Is a yes/no decision required? Or, are there alternatives to be evaluated and ranked? The final objective in a benefit-cost analysis and the corresponding form of the basic decision rule can significantly affect how much of a difference the variability in benefits estimates will make.

Early applications of benefit-cost analysis to water resource development projects required decision criteria that would be capable of ranking alternative investments.* The effect of benefits variability on these kinds of criteria--for example, net present value and internal rate of return--would be to complicate the analysis of the alternatives. The higher the variability, the more likely contradictory rankings would emerge across the same set of Projects. In addition, the need to compare across projects (or sites in our analysis) would likely add another source of variability, increasing the complexity of the task even more.

In the evaluation of environmental regulatory actions, the benefit-cost decision criterion is likely to be a yes/no type.† The focus of this kind of analysis is only on whether or not the net benefits are positive. The effect of benefits variability clearly differs in this case from those where the benefit-cost analysis is used to rank projects based on the size of the present value of net benefits or the internal rate of return. To some extent the yes/no decision framework reduces the impact of uncertainty. The regulatory action would be recommended whether net benefits are $100 or $1,000,000. In either case, the regulatory action results in an improvement over the status quo.

Clearly, the most difficult cases arise when the variability in benefits estimates leads to contradictory outcomes using the decision rule. The sup-

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*For example, see Eckstein [1958] and Krutilla and Eckstein [1958].

†This is not to suggest that benefit-cost analysis is a decision rule. It is not. Benefit-cost analysis provides a framework for organizing information that acknowledges the implications of an action for positive (or desired) outputs and its costs. To the extent that some outcomes are more difficult to value, the analysis provides a useful framework for pointing these out to the decisionmaker. For more details on this view of benefit-cost analysis, see Desvousges and Smith [1983].

port, or lack of support, for the decision based on only the benefit-cost analysis is much weaker. The policymaker could either use other criteria to support the decision or defer the decision until more is known.

8.4.2 Uncertainty in Benefit-Cost Analysis

Benefit variability is only one source of uncertainty in benefit-cost analysis. Uncertainty can also arise because the process that a policy attempts to influence is subject to random variation or because a nonstochastic process is incompletely observed (i.e., with error), or both. In the first case, random variations inherent in the process of interest would introduce uncertainty into the decision process, even for a policymaker with perfect ability to observe the process, to measure all aspects of the action's effects on it, and to know the "true" values of the outputs of the process. In the second case, which moves from this "ideal" conception of the relationship between policymaking and real world activities to one that corresponds more closely with actual conditions, additional sources of uncertainty are introduced. Here, not only is the process stochastic, but knowledge of the effects of policy actions is incomplete, the relationships between the outcomes and outputs that individuals value also are limited, and the values consumers place on these outputs are unknown. Each unknown dimension of the process must be estimated.

Many of these dimensions can affect the benefits attributable to actions and thereby lead to variability among estimates of their value, depending on the set of assumptions at each stage. This report has focused on one source of this variability--the measurement of an individual's valuation of one outcome of a policy (e.g., improved water quality). However, as Figure 1-1 illustrated, this is only one component of the full process.

The importance of this variability in estimates depends upon how it affects the process of using benefit-cost information in decisionmaking. Therefore, to consider the impact of benefit variability, it is essential also to consider the treatment of uncertainty in benefit-cost analysis generally and the specific aspects of benefit variability that are relevant to it.

How will the treatment of uncertainty in a benefit-cost assessment affect the treatment of benefits variability? Benefit-cost analysts have tried to address uncertainty with several different approaches:

- Adjusting the discount rate
- Using expected values for estimates benefits and costs
- Presenting interval estimates instead of point estimates and using extensive sensitivity analysis.

Obviously, the approach an analyst chooses has different implications for the evaluation of benefits variability. In the first approach, adjustments to the discounting rate attempt to account for an inherently stochastic process, and not necessarily to reflect estimation uncertainty--the primary source of benefits variability in our analysis. To use such adjustments in response to the type of benefits variability discussed in this analysis would mix issues associated with estimation uncertainty with those of selecting the appropriate rate to reflect society's time preference (e.g., the role of future generations).

In the second approach, probabilities are assigned to the various benefits and cost estimates to obtain the expected outcomes of each (see Haveman [1965] or United Nations [1972]). This approach is more general and could accommodate the type of uncertainty attributable to estimation. However, the exclusive reliance on the expected net benefits implied by this approach requires that the decisionmaker adopt a risk-neutral attitude. And, more importantly, it also requires that he be capable of estimating the likelihood of each set of estimates being realized. While this may seem straightforward, it is not given that the statistical properties of estimators are known. As Chapters 6 and 7 emphasize, estimates of demand models are only one of the elements entering a benefit appraisal. Benefits estimation usually involves many judgments, most of which are incompatible with the requirements of an expected value approach. For example, the range of benefit estimates within a given model specification can be evaluated, but assessing the appropriate range and assigning probabilities to each across models exceeds the scope of current methods.

In addition, the judgments required by the expected value approach do not end with the model. Benefits estimates are sensitive to the selection of a benefit measure and the features of both the recreation sites and their users. Since the specification of the probabilities for each set of benefit-cost estimates is crucial, this approach is not likely to provide an operational basis for these assignments. Nor is it likely to provide a basis for judging the importance of benefits variability.

The final approach for addressing uncertainty--using interval estimates and sensitivity analyses--is the most straightforward for handling uncertainty. In this approach, all estimates reflect their own levels of variability, and a sensitivity analysis can indicate the relative importance of each. Since this approach can be implemented, it is used in the hypothetical benefit-cost analyses below to judge the prospective importance of benefits variability within the framework of a set of hypothetical decisions for the Corps sites.

8.5 THE EFFECT OF BENEFITS VARIABILITY ON BENEFIT-COST ASSESSMENT

This section evaluates how the variability in benefits estimates affects the efficacy of conducting formal benefit-cost assessments. This evaluation uses the benefits estimates in developing several hypothetical benefit-cost analyses and then evaluates the effect of their variability on the final outcomes of the assessments.

8.5.1 Methodology for Hypothetical Assessments

The purpose of assessments is to gauge the effect that the variability in benefits estimates will have on the outcomes of a set of hypothetical decisions. In developing this appraisal, decisions are assumed to be yes/no regulatory judgments, rather than rankings of projects. With these assessments it will be possible to determine whether the outcomes remain unchanged with the value of the benefit estimates, i.e., whether all benefits estimates exceed cost or whether the range of benefits overlaps the assumed levels of cost.

The evaluations developed below would be more powerful if actual cost estimates of achieving various water quality levels at each site were available. However, in the absence of such data, these evaluations provide some insight of the effects of estimation uncertainty arising from the benefit side of a benefit-cost analysis. By omitting actual cost estimates and using assumed alternate levels of cost, an important source of variability is also eliminated: the variability in costs. Since cost estimation is beyond the scope of this research, it cannot be addressed here. However, it is an important one that is frequently deemphasized in the all-too-frequent controversy over benefits estimation.*

To evaluate the effects of variability, the hypothetical assessments consider five major elements:

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*It is important to remember the symmetry between benefits and costs. All that differentiates the two is the direction of the change from existing conditions. According to Portney [1984], the question of “costs” is an overlooked area by many environmental economists who are interested in benefit-cost analyses.

- Five assumed levels of annual costs--$1 million, $15 million, $25 million, $50 million and $100 million.

- One level of water quality--attaining fishable water quality.

- Two statistical estimation procedures--ML and OLS.

- Three measures of welfare change--two Marshallian and one Hicksian.

- User benefits are the exclusive focus.*

The basic data needed for these assessments are presented in Tables 8-1 and 8-2.

The last major element needed to evaluate variability is a translation of user benefits into an estimate of the total annual benefits of achieving fishable water quality. To obtain this estimate, the Corps of Engineers estimates of total recreation days at the site (from Table 4-3) are multiplied by the benefits per visitor day. This is somewhat similar to the unit-day approach to benefit estimation, which uses a standardized measure to value recreation activities. However, there are several important differences. First, the values are derived directly for each site from a generalized travel cost model. Second, the model adjusts for differences in characteristics of both sites and users. Finally, the values are based on the conventional willingness-to-pay criteria for measuring changes in well being.

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*Previous research, Desvousges, Smith, and McGivney [1983], has shown that for the Monongahela River the user benefits were approximately half the total benefits of improved water quality. Thus, this evaluation, which focuses on the user benefits from the travel cost model, omits an important source of benefits from the picture: intrinsic benefits--those not stemming from actual use of the recreation site.

In spite of these differences, however, this approach provides a rough approximation of benefits. For example, the measures of recreation and visitor days correspond in a limited sense. (See Loomis and Sorg [1982] for a discussion of this problem.) The Corps of Engineers procedure for classifying recreation days is incompatible with the visitor day measure because their procedure allocates days on a recreation activity basis. In this procedure, one visitor day in which the visitor both fishes and swims equals two recreation days. Ideally, one would have a measure of total visitor days derived from a statistical sample of visitors on an established accounting procedure for visitors such as the Forest Service 12 hour visitor day (see Loomis and Sorg [1982]). Since the purpose of the evaluation is primarily to illustrate the effect of variability, these specific limitations can be ignored.

8.5.2 Evaluation of Benefit Variability: The Results

Table 8-5 reports the annual benefits estimates that form the basis of the evaluation. As the ranges of estimated benefits for many of the 21 Corps of Engineers suggest, there is a substantial degree of variability among them for almost all the sites. Does this matter to the policy analyst who is trying to complete an assessment of benefits and costs? The answer is, "Yes." It does matter in many instances, but not enough to invalidate the analysis The outcomes reported in Table 8-6 illustrate this point. This table shows the total number of sites for which the variability in the benefits estimates creates contradictory outcomes for the analysis. A contradictory outcome occurs when the annual costs are within the range of the estimated benefits In other words, the benefit may either exceed or be less than the costs depending on either the estimator used in the demand analysis or the welfare measure.

Table 8-5 Annual Benefits of Fishable Water Quality

Site name	Site NO	Estimated annual benefits of attaining fishable water quality[a] ML M1	ML M2	ML H	OLS M1	OLS M2	OLS H
Arkabutla Lake, MS	301	16.1	30.4	57.3	153.5	101.9	222.8
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	0.4	1.2	2.4	6.1	3.3	25.6
Belton Lake, TX	304	1.0	2.2	4.2	37.0	24.4	63.2
Benbrook Lake, TX	305	0.3	2.3	4.7	24.8	44.2	48.6
Blakely Mt. Dam, Lake Ouachita, AR	307	0.6	0.5	2.2	4.1	4.7	12.2
Canton Lake, OK	308	0.3	1.2	2.1	37.4	9.9	330.0
Cordell Hull Dam and Reservoir, TN	310	6.0	4.8	16.3	23.8	18.9	53.1
DeGray Lake, AR	311	1.2	1.1	3.2	10.4	8.0	19.5
Grapevine Lake, TX	314	0.6	0.9	2.5	43.0	25.7	278.2
Grenada Lake, MS	316	5.1	8.8	14.2	83.6	42.4	86.9
Hords Creek Lake, TX	317	0.1	0.2	0.2	3.7	3.6	4.0
Melvern Lake, KS	322	0.4	1.0	2.2	18.2	9.7	11.0
Millwood Lake, AR	323	6.3	6.7	16.9	47.0	29.9	50.5
Mississippi River Pool NO. 6, MN	325	0.1	0.0	0.1	8.6	6.2	7.2
New Savannah Bluff Lock & Dam, GA	329	4.2	4.0	170.0	34.3	25.4	27.9
Ozark Lake, AR	331	5.8	1.2	6.3	16.3	15.9	42.3
Philpott Lake, VA	333	2.4	2.9	6.5	23.7	19.2	57.9
Proctor Lake, TN	337	0.6	0.0	0.1	9.3	3.5	4.0
Sam Rayburn Dam & Reservoir, TX	339	0.7	1.7	3.9	8.4	6.5	13.3
Sardis Lake, MS	340	1.2	1.4	2.6	24.9	17.8	44.3
Whitney Lake, TX	344	0.3	1.0	1.8	18.6	13.7	36.8

[a]Approximated by multiplying Corps of Engineers estimate (see Table 4-3) of number of recreation days by the visitor day estimates of consumer surplus (see Tables 8-1 and 8-2)

ML = Maximum likelihood estimator

OLS = Ordinary least-squares estimator

M1 = The Marshallian measure without an upper limit on the choke price.

M2 = The Marshallian measure with the maximum travel cost as a choke price

H = The Hicksian compensating surplus based on Hausman's quasi-expenditure function

There are several interesting implications for policy analysis that can be drawn from the information in Table 8-6. As we might expect, since the cost figure is treated as a series of equally plausible alternatives, the total number of contradictory outcomes is very sensitive to what is assumed to be the level of costs. The fewest contradictions occur at the $100 million annual cost level, with only four cases yielding contradictory outcomes. The remaining 17 sites all show benefits less than costs. Of the four contradictory cases, three of them have only one benefit estimate (the Hicksian measure using OLS) that exceeded the costs.

For an assumed level of costs of $50 million annually, nine sites yielded contradictory outcomes. In this case, seven of the nine sites had only one benefits estimate in excess of the cost. The highest estimate, again, was the Hicksian measure based on the OLS estimator in the demand analysis.

At the three lowest assumed levels of cost, the variability in benefits estimates has the most apparent impact. In the worst case, annual costs of $10 million, 17 out of the 21 sites yielded contradictory outcomes and had multiple benefit estimates on either side of the assumed cost. Thus, for costs in the range of $1 million to $25 million annually, the variability in the benefits estimates substantially complicates the task for the policy analyst. No clearcut assessment of benefits versus cost emerges in these cases. The outcome is very sensitive to either the estimator or the welfare measure. In some cases these factors compound the degree of variability. This evaluation has implicitly viewed the benefits estimates equally.

Should the analyst attach more weight or have more confidence in one estimator or the other, in one welfare measure or the other? The answer to both of these questions is a qualified "No." As noted in Chapter 6, neither

Table 8-6. Sites with Contradictory Outcomes

Site number	Total	Assumed annual cost level ($\$10^6$)
302, 305, 307, 308, 314, 317, 322, 325, 337, 339, 344	11	1
302, 304, 305, 307, 308, 310, 311, 314, 316, 322, 323, 329, 331, 333, 339, 340, 344	17	10
301, 302, 304, 305, 308, 310, 314, 316, 323, 329, 331, 333, 340, 344	14	25
301, 304, 308, 310, 314, 316, 323, 329, 333	9	50
301, 308, 314, 329	4	100

the ML nor OLS estimates can be regarded as the true values. Both require assumptions that limit their ability to be considered as the truth. On an intuitive basis, one might attach more weight to the ML estimates because they reflect the character of the data used in the model. However, this preference is based largely on concern over bias in the estimates of the parameters rather than in the spread or variance in their sampling distributions.

In the case of the welfare measures, one, in principle, should prefer the Hicksian measure that holds utility constant over the Marshallian measures. However, the empirical approximations necessary to obtain the Hicksian measure (see Chapter 7) reduce the advantage of the Hicksian measure that stems from theory. Thus, there is again no truth, no Greenwich mean time, against which the empirical estimates of welfare change can be compared.

Is this type of variability unique to this study? While a definitive answer to this question is difficult, the experience of Vaughan and Russell [1982a] suggests that the answer is probably "No." They found that sample design, methodology, variable specification, and valuation assumptions all made substantial differences in the final benefit estimates, with the ratio of highest to lowest of about 3 to 1. The ratio increased to 10 to 1 when they examined the sensitivity across valuation assumptions in addition to the other factors. Some caution is required in drawing inferences across studies because the Vaughan-Russell objective was to develop national benefit estimates, while all the estimates in this study are for individual sites. Because differences in site or user characteristics are not a source of variability in our study, one would expect them to be lower. The General Accounting Office [1984] also has noted there is a substantial variation in benefit (and cost) estimates in several EPA regulatory impact analyses that they reviewed. They suggest that, although high variability levels may complicate the analysis of benefits and costs, the assessment is still better than dealing with the uncertainty caused by no information.

8.6 COMPARISON OF BENEFITS ACROSS STUDIES

This section compares the benefits estimates from this study with those of Vaughan and Russell [1982a] and Loomis and Sorg [1982]. However, before turning to the comparison, the key features of each of the studies that are important to the comparison are summarized.

8.6.1 Vaughan and Russell Study

The Vaughan and Russell [1982a] study, the most ambitious and detailed application of a recreation participation model to date, focuses on the recreational fishing benefits that arise from a change in water quality. It uses the fact that the more “desirable” freshwater sport fish--coldwater and certain warm-water species--require better water quality. Improved water quality may alter the types of fish that can be supported in a water body. Assuming the supporting recreation facilities are available, Vaughan and Russell suggest that there will be a change in the type of fish (and perhaps a net increase in the level of fishing participation) from less desirable to more desirable varieties. The sources of benefits from the water quality change arise from:

- The change in the composition of fishing activities.
- Any net increase in the level of participation in fishing.

Other key features of the Vaughan-Russell study include:

- An approach that combines participation and travel cost models to predict increased fishing and to value the increase.
- A travel cost model that includes site characteristics affecting fish species but not water quality.

- Several alternative model specifications to evaluate different treatments of travel costs.

- Travel cost data from fisheries that emphasized bag-limit fishing and rather ordinary environmental settings.

8.6.2 Loomis and Sorg Study

The Loomis and Sorg [1982] study was an extensive critical review of the existing literature on valuing different activities. Their main objective was to develop consensus estimates of the values of different recreation activities on National Forest lands. Key features include:

- Estimates are gleaned from both travel cost and contingent valuation studies of varying levels of quality.

- Estimates are adjusted to standardize the treatments of travel time, substitute sites, protest bids, and several other factors. Travel time is valued implicitly as a fraction of the wage rate with estimates omitting travel time costs inflated by 30 percent.

- Benefits estimates are lowered from travel cost studies using individual observation data.

- Activities are the main focus; therefore, most studies did not use models that linked water quality changes directly to the model.

8.6.3 The Present Study

For purposes of comparison, the key features of this study are:

- Uses a generalized travel cost model that explicitly included water quality as a site characteristic.

- Estimated the model using both OLS and ML estimation procedures.

- Employed a hedonic wage model to predict wage rates to value the opportunity cost of time.

- Estimated the travel cost model using observations on individual's visits to sites.

- Included sites that supported a diverse array of recreation sites.

- Estimated changes in welfare using both Marshallian and Hicksian welfare measures.

Table 8-7 compares the benefits estimates from the generalized travel cost model with estimates from Vaughan-Russell [1982a] and Loomis-Sorg [1982]. Both of these studies derive benefit estimates for water quality improvements based on the value of fishing experiences with cold-water species as the increment over warm-water species. The remaining rows in the table report the predictions for Monongahela River sites from the generalized travel cost model (based on the OLS estimates) in Desvousges, Smith, and McGivney [1983],* the same model used for predictions for the 21 Army Corps of Engineers sites and average socioeconomic characteristics of users, and the ML estimates of this model for the same group. Only the Marshallian measure (M2) is shown in the table because it corresponds more closely to the other two studies.

While the benefit estimates derived from the OLS estimated version of the generalized travel cost model were clearly in the range of estimates from the other studies using the Monongahela survey data, they are not consistent when the data on the 21 Corps of Engineers sites are used. Only estimates at the lowest end of the range are comparable to the highest estimates from Vaughan-Russell [1982a]. If either M1 or the Hicksian measure were considered, the estimates from the OLS model are considerably larger than the values from other studies. The substantially lower values of water quality improvements for the Corps sites with the ML estimator fall within the range in both the Vaughan-Russell [1982a] and Loomis-Sorg [1982] studies.

Comparing the results on a per-day basis for all three studies yields similar conclusions. In the case of the Monongahela River sites, nearly all trips

*These values were estimated for users of the Monongahela River in a household survey of five counties in the river basin.

Table 8-7. A Comparison of the Estimates of the Benefits of Water Quality improvements

	Original estimate	1982 dollars
Vaughan-Russell [1982a]	$4.00 to $8.00 per person per day, 1980 dollars; range over model used	$4.68-$9.37
Loomis-Sorg [1982]	$1.00 to $3.00 per person per day; 1982 dollars, range over region	$1.00-$3.00
Desvousges, Smith, and McGivney [1983]	$0.98 to $2.03 per trip, 1981 dollars, for Monongahela sites (boatable to fishable)	$1.04-$2.15
Generalized travel cost model--OLS estimates	$5.87 to $54.20[a] per trip ($2.24 to $122.00 per visitor day) in 1977 dollars for the Corps sites (boatable to fishable water quality change)	$9.35-$86.34 ($3.57-$194.35)
Generalized travel cost model--ML estimates	$0.08 to $6.19 per trip ($0.04 to $19.00 per visitor day) in 1977 dollars for the Corps sites (boatable to fishable water quality change)	$0.13-$9.86 ($0.06-$30.27)

[a] These estimates relate only to the Marshallian consumer surplus (M2). Comparable estimates can be derived for the other benefit measures in Table 7-5 using the scaling factor 1.593 to convert from 1977 to 1982 dollars. The benefits per visitor day are shown in parentheses.

were 1-day trips, so in this case, there is no distinction. However, this is not the case for the 21 Corps of Engineers sites where the average number of days per trip ranged from 3 hours at the New Savannah Bluff Lock and Dam to over 5 days at Lake Ouachita. The OLS-based estimates on a per-day basis have all but two extreme values within the range of estimates, while the ML-based estimates fall at the lower end of the existing range with the exception of the New Savannah Bluff Lock and Dam site.

There are several possible explanations for the relatively larger benefits estimates from this study. First, our measure of travel cost used the full wage rate to value the opportunity cost of travel time. The Loomis-Sorg estimates used a fraction of the wage rate while only the higher estimates in Vaughan-Russell used the full cost of time. Several studies (Cesario [1976], Cesario and Knetsch [1970]) have shown that the benefit estimates derived from the travel cost model are very sensitive to the treatment of time. Are any of the treatments more correct? Recent evidence in Smith, Desvousges, and McGivney [1983] suggests that given data limitations in most recreation studies all treatments of travel time can best be viewed as approximations of the true opportunity cost. Because of limited differences in the empirical evidence for different treatments of travel time, we have argued for using the full wage rate to value travel time. Disagreements in the treatment of travel time in travel cost models remain an important source of variation in benefit estimates across studies.

A final source of difference between our estimates and those from Vaughan-Russell is in the nature of the recreation sites. For example, Vaughan-Russell [1982a, p. 175] state:

> Our estimates . . . . are based on data from a sample of fishing sites which turned out to have many members with characteristics quite different from those of the free, public, and largely natural sites we usually have in mind when we speak of recreational fishing. The surveyed sites appeared to stress bag and to provide an ambience in which the hand of man was not hidden.

While the Corps of Engineers sites in our study are not exempt from the hand of man, the characteristics of many of the sites (described in Chapter 4) are far more representative of recreational fishing. Our larger estimates might be consistent with the differences in sites but are probably lower than estimates of the value of fishable water quality in the unique natural environments that Vaughan and Russell allude to in their appraisal.

In summary, our estimates on the value of improving water quality are generally larger than those from other studies. The estimates based on the ML estimator are much closer to the others while the OLS values tend to be much larger. The reasons for these differences can be traced to differences in data, variable measurement and model specification between the studies.

8.7 RECOMMENDATIONS FOR BENEFITS TRANSFER

This section discusses the issue of benefits transfer and provides some recommendations for transferring the benefits estimates from this study to other studies. Benefits transfer refers to the process of using estimates of the benefits associated with an improvement in some aspect of environmental quality, such as water quality, that have been derived for one location or subset of the population in another context. This other context may be another site that will experience the improvement under the policy to be evaluated. Or it could be that all individuals in the population will experience the same change as the sampled group. The need to consider this task arises because policy analysts frequently must rely on available research to evaluate potential actions. There is usually insufficient time to develop the needed information to estimate the benefits for the specific site or population under study.

8.7.1 Current Practice for Benefits Transfer

When there is substantial variability in benefit estimates, as our results indicate, then the current practices for applying benefit estimates in a variety of contexts with limited adjustments may need to be reconsidered. Nevertheless, the issue of benefits transfer is one that is largely ignored in the literature on benefits research. Unfortunately, the decisions faced by policy analysts of trying to adapt benefits estimates from one study to another remain in the province of obscured appendixes. Freeman [1979, 1984] has discussed some of the aspects of transferring estimates of national benefits from one study to another. Some of the major problems that arise in transferring benefits estimates from one site to another include:

- Site-specific differences in the technical effects of the regulation (e.g., which water quality parameters change), the responses of economic entities (e.g., fishermen may not change their pattern of behavior), and other characteristics (e.g., poor access to the site may limit benefits).

- Differences in the characteristics of the population of individuals or firms that affect their behavioral responses (e.g., the primarily high income visitors to Mississippi River Pool Number 6 that affect their demand for the site's services).

- Differences in the time periods between the studies (travel cost demand models estimated with 1977 data may be poorly suited to predict behavior in 1984 at the same site much less other sites).

- Limited data for the site under consideration for the transfer.

In light of these problems, the benefits transfer task facing the policy analysis may appear insurmountable. These difficulties may, in part, explain the emphasis of the 1982 Principles and Guidelines on site-specific analyses using either the travel cost or contingent valuation approaches. While these guidelines make imminent sense for water projects involving large expenditures of public funds, there may be a middle ground for transferring benefits from one site to another to develop an approximate or first-cut set of estimates for evaluating the effects of a regulatory action. Depending on the outcome of this evaluation, the analyst could decide if more detailed analysis are warranted.

8.7.2 General Recommendations for Benefits Transfer

Based on the experience gained with the travel cost model in this study, several general suggestions for benefits transfer can be made. However, these suggestions are limited by the scope of our analysis--i.e., by the fact that it did not include the contingent valuation approach or attempt to measure intrinsic benefits. Nevertheless, for transferring user benefits of water quality improvements derived from the generalized travel cost model, we recommend the following:

- Match the characteristics of sites and users as closely as possible. This study has shown the substantial effects differences these factors can have on benefit estimates.
- Establish linkages, to the extent possible, between regulations, water quality changes, and changes in individual's decisions.
- Use estimates based on models that have linked water quality to changes in individual behavior.
- Use the model's coefficients with the data on the site to be valued to predict benefits. Better predictions should result if differences in site and user characteristics can be accounted for.
- Evaluate whether the site to be valued lies within the range of predictions (and data) on the selected model.
- Use interval estimates for all facets of analysis.
- Evaluate the sensitivity of model's coefficients to type of estimation procedure employed.

- Evaluate the sensitivity of benefits estimates to the type of welfare measure employed with the model.
- Conduct sensitivity analyses of all key assumptions.
- Identify sources of benefits omitted from the analysis.* (For example, travel cost models do not measure intrinsic benefits.)

These recommendations are not a panacea for addressing the transfer of benefits. They are only an attempt to provide some perspective on the question based on the research conducted in this study. Additional discussion on basic issues in benefit-cost analysis and water quality programs can be found in Desvousges and Smith [1983]. Clearly, these recommendations have just scratched the surface of an important and complex question.

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*Or consider following the recommendations of Fisher and Raucher [1984] for approximating intrinsic benefits based on estimates of user benefits.

CHAPTER 9

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Appendix A

The Varying Parameter Model: Some Background

APPENDIX A

THE VARYING PARAMETER MODEL: SOME BACKGROUND

The varying parameter model was first proposed by Freeman [1979] as a potentially attractive framework for estimating the implications of site characteristics, like water quality, for recreation site demand. Since that time, two variations of the model have been used in two different applications: Vaughan and Russell [1982a] used it in their analyses of the demand for sites providing opportunities for freshwater fishing, and Desvousges, Smith, and McGivney [1983] used a two-step form of the model to estimate a varying parameter demand model for water-based recreation sites.

The general form of the first stage of the Desvousges, Smith, and McGivney [1983] model is shown in Equation (A.1):

$$\ln V = a + bTC + cY , \qquad (A.1)$$

where

$\ln V$ = the natural log of the number of visits by a household to the site in a recreation season

TC = the travel cost per visit to the site, including out-of-pocket vehicle operating costs and the opportunity cost of the time spent traveling

Y = family income.

This stage of the model assumes that variation in the estimates of a, b, and c across sites reflects the effects of site characteristics on the representative individual's demand for services from each of the sites. Thus, each estimate provides the basis for describing how a change in any site attribute would affect demand for that site.

The second stage of the model involves estimating the relationship between variations in the site-specific estimates of a, b, and c and each site's attributes. Since precision in the estimates of demand parameters a, b, and c in Equation (A.1) varies, Desvousges, Smith, and McGivney [1983] used a generalized least-squares (GLS) procedure to account for the quality of the estimates.

Although the two variations of the generalized travel cost model follow the same lines, there are some differences. In particular, the Vaughan-Russell [1982] model was implemented in one step with a GLS estimator used to account for the heteroscedasticity implied by the model's error structure. This arrangement results in equivalent results when the site demand model and the models linking site characteristics to these demand parameters are all known a priori.* In practice, however, it is unlikely that the forms for all of these functions will be known in advance. For example, it is common practice to find substantial pretesting and model evaluation in recreation modeling.† Under these conditions, the two-step approach makes this evaluation process easier to interpret. Evaluation is more complicated in the one-step approach because the effects of both the site demand model specifications and site characteristics model interact to affect the estimated results.

*See Saxonhouse [1977] for a formal proof of this result. It should be noted that it relies on the assumption that both approaches describe the error structure involved in an identical format.

†See Desvousges, Smith, and McGivney [1983] Appendix G for a description of the models considered prior to the forms used for the development of the generalized travel cost model. A good description of the literature to the late seventies can be found in Wallace's survey [1977]. A more recent appraisal of the issues with a somewhat different perspective on the implications of this literature is found in Judge and Bock [1983].

To illustrate this point, consider an example. Assume the travel cost demand function is semi-log arithmic, with price the only determinant of the quantity demanded, Q, as in Equation (A.2) below:

$$\ln Q = \alpha_0 + \alpha_1 P + \varepsilon \ . \tag{A.2}$$

If we assume that each parameter is a nonstochastic function of two attributes, Z_1 and Z_2, as in Equations (A.3) and (A.4), we can derive Equation (A.5) by substituting (A.3) and (A.4) into (A.2):

$$\alpha_0 = a_0 + a_1 Z_1 + a_2 Z_2 \tag{A.3}$$

$$\alpha_1 = b_0 + b_1 Z_1 + b_2 Z_2 \tag{A.4}$$

$$\ln Q = a_0 + a_1 Z_1 + a_2 Z_2 + b_0 P + b_1 P Z_1 + b_2 P Z_2 + \varepsilon \ . \tag{A.5}$$

Now consider the prospect of adding an additional determinant to each equation and judging whether it should be treated as a likely influence on the dependent variables in each case. Suppose the additional variables were Y and Z_3. If Y entered Equation (A.2) and its parameter was also affected by Z_1, Z_2, and Z_3, we would have Equation (A.6) (assuming α_0 and α_1 were also affected by Z_3):

$$\begin{aligned} \ln Q = {} & a_0 + a_1 Z_1 + a_2 Z_2 + a_3 Z_3 + b_0 P + b_1 P Z_1 + b_2 P Z_2 \\ & + b_3 P Z_3 + C_0 Y + C_1 Y Z_1 + C_2 Y Z_2 + C_3 Y Z_3 + \varepsilon \ . \end{aligned} \tag{A.6}$$

The more variables introduced at either stage, the greater the expansion in the specified “final” demand model and the greater the prospects for an increase in collinearity. This collinearity in regressors can make the prospects for discriminating among models at each stage especially difficult: Vaughan and Russell [1982] had a limited number of observations on the use of each site, so splitting the estimation process was not a practical alternative. By contrast, the Federal Estate Survey did permit estimation Of each Site demand individually.

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Appendix B

Benefit Estimates for Activity Mix Scenarios

APPENDIX B

BENEFIT ESTIMATES FOR ACTIVITY MIX SCENARIOS

This appendix describes the benefit estimates associated with four scenarios for water quality/activity/congestion combinations. As noted in the text for the definition of a consistent aggregate site demand, the activity mix must remain constant irrespective of the change in water quality. The benefit estimates reported in Chapter 6 were for the existing activity and site characteristics. Tables B-1 and B-2 report estimates for the sites when: (1) the activities are assumed to be exclusively boating with its activity index set at unity and the fishing index at zero, and (2) there is assumed to be an equal mix of boating and fishing activities with each of the indexes set to one-half. For each of these cases we considered circumstances with congestion during peak periods and no congestion.

Unfortunately, as the text indicated, the results do not appear to conform to our a priori expectations and are difficult to interpret in relation to other studies that have attempted to value specific activities.*

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*As an example, see Loomis, John, and Cindy Sorg, 1982, "A Critical Summary of Empirical Estimates of the Values of Wildlife, Wilderness and General Recreation Related to National Forest Regions," U.S. Forest Service, Denver, Colorado, 1982.

TABLE B-1. ALL BOATING ACTIVITIES SCENARIO
(exclusively boating)

Site name	Site No.	Congested in peak: Boatable to fishable M	Congested in peak: Boatable to fishable H	Congested in peak: Boatable to swimmable M	Congested in peak: Boatable to swimmable H	Noncongested in peak: Boatable to fishable M	Noncongested in peak: Boatable to fishable H	Noncongested in peak: Boatable to swimmable M	Noncongested in peak: Boatable to swimmable H
Arkabutla Lake, MS	301	15.66	35.34	36.18	89.54	2.35	8.78	5.35	21.92
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	5.46	14.32	12.54	35.34	2.41	9.21	5.53	22.67
Belton Lake, TX	304	7.93	19.57	17.83	49.63	0.83	3.96	1.86	9.94
Benbrook Lake, TX	305	1.57	4.36	3.45	10.93	0.22	1.24	0.48	3.11
Blakely Mt. Dam, Lake Ouachita, AR	307	8.15	28.79	18.57	73.98	0.52	3.56	1.17	9.00
Canton Lake, OK	308	20.40	66.10	45.46	186.51	2.74	11.27	6.07	36.03
Cordell Hull Dam and Reservoir, TN	310	5.18	10.85	11.74	27.39	0.71	2.63	1.59	6.60
DeGray Lake, AR	311	1.61	8.67	3.59	22.08	0.13	1.42	0.28	3.59
Grapevine Lake, TX	314	1.42	3.79	3.13	9.57	0.15	0.83	0.33	2.07
Grenada Lake, MS	316	37.95	59.30	90.89	151.63	4.90	10.91	11.52	27.35
Hords Creek Lake, TX	317	0.19	0.61	0.42	1.53	0.02	0.14	0.05	0.36
Melvern Lake, KS	322	0.81	3.01	1.80	7.55	0.12	0.84	0.26	2.11
Millwood Lake, AR	323	373.60	618.13	897.57	1,697.90	4.81	18.78	10.78	47.89
Mississippi River Pool No. 6, MN	325	1.15	4.38	2.55	12.91	0.09	0.48	0.19	1.80
New Savannah Bluff Lock & Dam, GA	323	0.30	1.30	0.67	3.24	0.08	0.58	0.19	1.44
Ozark Lake, AR	331	0.15	0.55	0.34	1.40	0.01	0.07	0.03	0.17
Philpott Lake, VA	333	5.07	22.91	11.51	99.74	0.91	1.71	2.05	4.26
Proctor Lake, TN	337	1.20	16.05	2.70	4.94	0.11	0.35	0.26	0.89
Sam Rayburn Dam & Reservoir, TX	333	203.68	1,213.11	463.45	3,171.86	5.70	73.73	12.72	188.48
Sardis Lake, MS	340	3.36	7. 73	7.75	19.57	0.43	1.61	0.98	4.04
Whitney Lake, TN	344	17.95	37.94	40.73	96.79	1.48	6.45	3.27	16.22

M = The Marshallian measure with the maximum travel cost as a choke price.

H = The Hicksian compensating surplus based on Hausman's quasi-expenditure function.

TABLE B-2. BOATING-FISHING ACTIVITIES SCENARIO
(equal mix of boating and fishing)

		Congested in peak				Noncongested in peak			
		Boatable to fishable		Boatable to swimmable		Boatable to fishable		Boatable to swimmable	
Site name	Site No.	M	H	M	H	M	H	M	H
Arkabutla Lake, MS	301	9.67	13.60	22.01	32.41	2.13	3.45	5.28	7.93
Lock and Dam No. 2 (Arkansas River Navigation System), AR	302	7.62	9.65	18.12	23.03	3.77	6.91	8.99	16.55
Belton Lake, TX	304	5.47	6.67	12.19	15.15	0.83	1.14	1.85	2.57
Benbrook Lake, TX	305	2.01	2.35	4.53	5.40	0.34	0.45	0.76	1.02
Blakely Mt. Dam, Lake Ouachita, AR	307	2.76	3.18	6.06	7.02	0.31	0.65	0.68	1.43
Canton Lake, OK	308	19.42	20.84	43.49	46.94	3.16	3.42	7.12	7.75
Cordell Hull Dam and Reservoir, TN	310	3.89	4.54	8.73	10.36	0.70	0.87	1.58	1.98
DeGray Lake, AR	311	1.01	1.42	2.21	3.13	0.11	0.32	0.25	0.70
Grapevine Lake, TX	314	1.33	1.58	2.95	3.57	0.17	0.23	0.39	0.51
Grenada Lake, MS	316	12.07	14.85	27.35	34.20	2.91	3.55	6.61	8.08
Hords Creek Lake, TX	317	0.15	1.54	0.33	0.34	0.02	0.04	0.05	0.09
Melvern Lake, KS	322	0.92	1.72	2.08	4.42	0.16	0.30	0.35	0.67
Millwood Lake, AR	323	-17.40	-7.65	-17.25	2.30	2.63	[a]	5.84	6.34
Mississippi River Pool No. 6, MN	325	0.73	0.82	1.59	1.79	0.07	0.08	0.15	0.17
New Savannah Bluff Lock & Dam, GA	329	0.35	0.63	0.81	1.47	0.11	0.31	0.25	0.72
Ozark Lake, AR	331	0.04	0.06	0.08	0.13	0.00	0.01	0.01	0.02
Philpott Lake, VA	333	4.46	4.81	10.10	10.99	0.96	1.08	2.19	2.49
Proctor Lake, TN	337	0.77	0.88	1.68	1.94	0.09	0.12	0.19	0.28
Sam Rayburn Dam & Reservoir, TX	339	-10.98	-29.88	28.27	56.91	2.42	8.28	5.34	18.30
Sardis Lake, MS	340	1.74	2.73	3.88	6.48	0.29	0.45	0.65	1.00
Whitney Lake, TX	344	10.95	12.94	24.33	29.10	1.48	1.66	3.29	3.17

[a] Estimated value could not be interpreted.

M = The Marshallian measure with the maximum travel cost as a choke price.

H = The Hicksian compensating surplus based on Hausman's quasi-expenditure function.

Appendix C

Regression Results for Activities as Functions of Site Characteristics

APPENDIX C

REGRESSION RESULTS FOR ACTIVITIES AS FUNCTIONS OF SITE CHARACTERISTICS

The purpose of this appendix is to provide a brief summary of the regression results that related the recreation activities to site characteristics. These tables provide a rough appraisal of the potential for simultaneity between site characteristics and site activities. None of the results suggest strong relationships, but this may be more a function of deficiencies in our activity measures than the lack of a valid relationship.

TABLE C-1. ACTIVITY AND SITE CHARACTERISTICS REGRESSIONS

	Measure of boating	Measure of fishing
Intercept	3.7×10^{-1} (1.814)	7.3×10^{-1} (4.443)
SHORMILE	1.6×10^{-4} (0.755)	3.4×10^{-5} (0.195)
ARSIZE	3.2×10^{-1} (2.121)	1.8×10^{-1} (1.495)
DOM	-1.9×10^{-3} (-1.544)	-9.9×10^{-4} (-0.957)
DOV	7.8×10^{-6} (1.640)	1.2×10^{-6} (0.339)
COLD	9.3×10^{-2} (1.238)	-2.5×10^{-2} (-0.417)
STOCK	-6.5×10^{-2} (0.676)	-1.8×10^{-1} (-2.272)
SUB	6.7×10^{-2} (0.064)	-5.0×10^{-2} (-0.961)
R^2	0.39	0.30
$\bar{R}^2$	0.21	0.09
D.F.	30	30

SHORMILE = Total shoremiles at the site during peak visitation period.

ARSIZE = Total water area plus total land area.

DOM = Mean level of dissolved oxygen.

DOV = Variance in dissolved oxygen.

COLD = Qualitative variable indicating presence of coldwater gamefish (=1).

STOCK = Qualitative variable indicating presence of an on-going fish-stocking program (=1).

SUB = Qualitative measure of substitute.